AD-A098 271 HARPER-OWES CO SEATTLE WA F/G 8/8

DUWAMISH WATERWAYS NAVIGATION IMPROVEMENT STUDY: ANALYSIS OF IM--ETC(U)

FEB 81 DACW67-80-C-0128

UNCLASSIFIED NL

1 OF 1

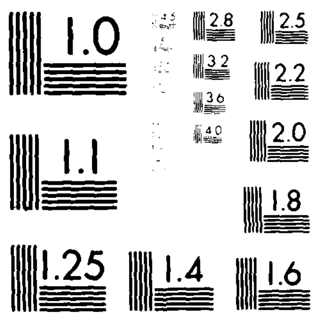
AD A 098271

END

DATE FILMED

5-81

DTIC



MICROCOPY RESOLUTION TEST CHART

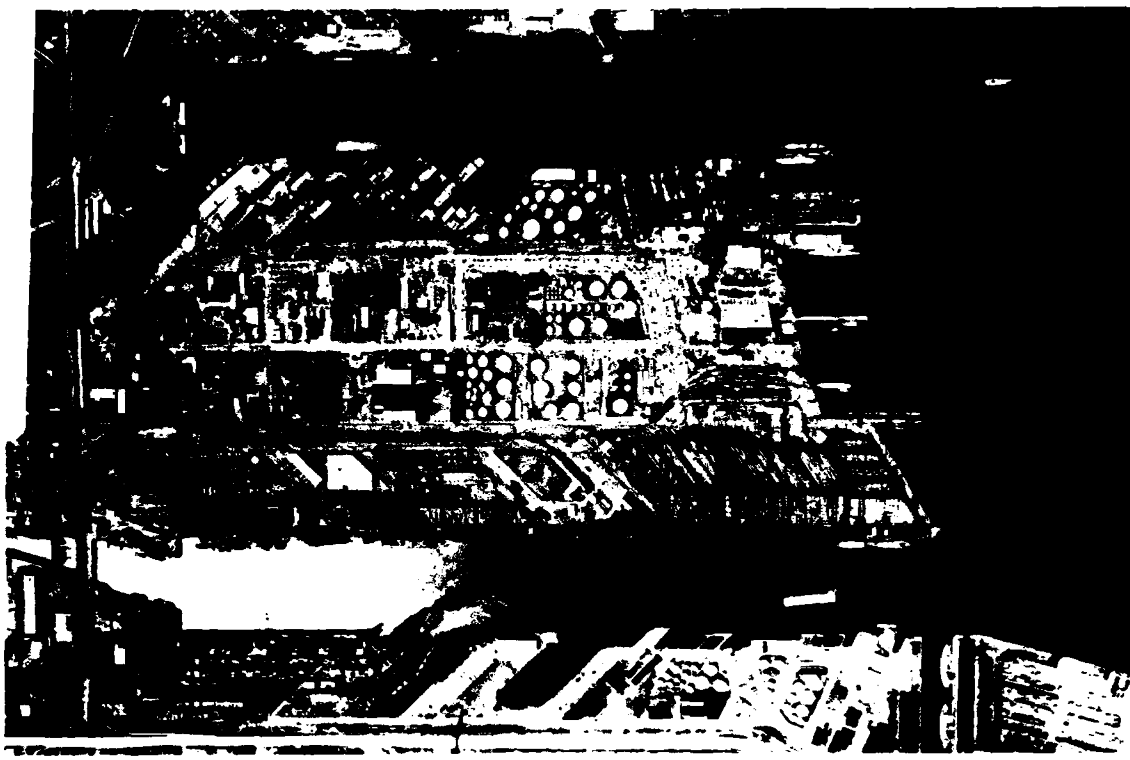
NATIONAL BUREAU OF STANDARDS-1963-A

15

DUWAMISH WATERWAYS NAVIGATION IMPROVEMENT STUDY: (3)

Analysis of Impacts on Water Quality and Salt Wedge Characteristics.

LEVEL III



PREPARED BY
HARPER-OWES

FOR

U.S. ARMY CORPS OF ENGINEERS
SEATTLE DISTRICT

FEBRUARY 1981

EAST/WEST DUWAMISH WATERWAY NAVIGATION IMPROVEMENT STUDY

Cover Photograph: The aerial photograph of Harbor Island and the East and West Duwamish Waterways depicts the intense industrial and port related activities in the area. Harbor Island, created by earlier disposal of dredged materials, is located at the mouth of the Duwamish estuary where the estuary meets Elliott Bay. Photograph by Seattle District, Corps of Engineers.

SECURITY CLASSIFICATION OF THIS PAGE (When Data Entered)

REPORT DOCUMENTATION PAGE

READ INSTRUCTIONS BEFORE COMPLETING FORM

1. REPORT NUMBER

2. GOVT ACCESSION NO. AD-A098271

3. RECIPIENT'S CATALOG NUMBER

4. TITLE (and Subtitle)

Duwamish Waterway Navigation Improvement Study: Analysis of Impacts on Water Quality and Salt Wedge Characteristics.

5. TYPE OF REPORT & PERIOD COVERED

Final rept.

6. PERFORMING ORG. REPORT NUMBER

7. AUTHOR(s)

Harper-Owes Company

8. CONTRACT OR GRANT NUMBER(s)

DACW67-80-C-0128

9. PERFORMING ORGANIZATION NAME AND ADDRESS

301 Commuter Building
65 Marion Street
Seattle, Washington 98104

10. PROGRAM ELEMENT, PROJECT, TASK AREA & WORK UNIT NUMBERS

11. CONTROLLING OFFICE NAME AND ADDRESS

U.S. Army Corps of Engineers
Seattle District, P.O. Box C-3755
4735 E. Marginal Way S., Seattle, WA 98124

12. ~~REPORT DATE~~

Feb[redacted]81

13. NUMBER OF PAGES

90

14. MONITORING AGENCY NAME & ADDRESS (if different from Controlling Office)

92

15. SECURITY CLASS. (of this report)

Unclassified

15a. DECLASSIFICATION/DOWNGRADING SCHEDULE

16. DISTRIBUTION STATEMENT (of this Report)

Approved for Public Release; Distribution Unlimited

17. DISTRIBUTION STATEMENT (of the abstract entered in Block 20, if different from Report)

Approved for Public Release; Distribution Unlimited

18. SUPPLEMENTARY NOTES

19. KEY WORDS (Continue on reverse side if necessary and identify by block number)

WATER QUALITY
DREDGING IMPACTS
DUWAMISH RIVER
DISSOLVED OXYGEN
SALT WEDGE
SEDIMENT OXYGEN DEMAND

20. ABSTRACT (Continue on reverse side if necessary and identify by block number)

A review of historical water quality data and studies in the Duwamish Estuary has revealed that several changes in water quality conditions have occurred since the mid-1960's. Surface water D.O. concentrations have dropped near the head of navigation in response to nitrification of increased ammonia discharged from the Renton Wastewater Treatment Plant (RTP). However, salt-wedge D.O. concentrations, which historically have been depressed to 1 mg/l, have increased markedly following sewage diversion from the estuary in the late 1960's. This improvement was found to be principally due to decreased oxygen consumption in the wedge. Based on a cont.

DD FORM 1473, 1 JAN 73 EDITION OF 1 NOV 65 IS OBSOLETE

SECURITY CLASSIFICATION OF THIS PAGE (When Data Entered)

20. (Cont)

conts

comparison of total saltwater wedge consumption (calculated) and sediment oxygen demand (SOD) measurements taken during August 1973, wedge D.O. uptake was found to be almost soley benthic (bottom-related). Other factors influencing wedge D.O. (freshwater input, tidal exchange and seawater inflow concentration) have not changed significantly since 1966.

Since oxygen demand is primarily benthic, a dredging project which only increases wedge depth (area constant) will have minimal effect on wedge D.O. concentrations because the longer residence (reaction) time will be offset by the increased dilution of SOD. The alternatives considered will have negligible impact on phytoplankton production. For the mid-sized alternative, wedge D.O. concentrations are expected to change by less than 0.1 mg/l. The proposed dredging project is not expected to enhance problems associated with ammonia toxicity and toxin spills in the estuary.

DUWAMISH WATERWAY NAVIGATION IMPROVEMENT STUDY:

ANALYSIS OF IMPACTS ON WATER QUALITY AND SALT WEDGE CHARACTERISTICS

Prepared By

HARPER-OWES

for

U. S. ARMY CORPS OF ENGINEERS

SEATTLE DISTRICT

Contract No. DACW 67-80-C-0128

February 1981

TABLE OF CONTENTS

EXECUTIVE SUMMARY

A review of historical water quality data and studies in the Duwamish Estuary has revealed that several changes in water quality conditions have occurred since the mid-1960's. Surface water D.O. concentrations have dropped near the head of navigation in response to nitrification of increased ammonia discharged from the Renton Wastewater Treatment Plant (RTP). However, salt-wedge D.O. concentrations, which historically have been depressed to 1 mg/l, have increased markedly following sewage diversion from the estuary in the late 1960's. This improvement was found to be principally due to decreased oxygen consumption in the wedge. Based on a comparison of total saltwater wedge consumption (calculated) and sediment oxygen demand (SOD) measurements taken during August, 1973, wedge D.O. uptake was found to be almost solely benthic (bottom-related). Other factors influencing wedge D.O. (freshwater input, tidal exchange and seawater inflow concentration) have not changed significantly since 1966.

Present major sources of organic enrichment to the saltwater wedge include RTP discharge and natural/background upstream contributions. Minor sources include combined sewer overflow discharges and phytoplankton. Phytoplankton blooms are now less frequent and less intense that in the late 1960's, apparently because of reductions in upstream inoculum (algae seed) and/or increased toxicity of RTP effluent. Major sources of inoculum may have been the Kent and Auburn lagoons, which were eliminated in the early 1970's. Short-term limitation of earlier (1966-67) blooms was due, in part, to CO_2 limitation. Phytoplankton blooms previously contributed to short-term oxygen depressions and increased the toxicity of ammonia by raising pH. Recurrence of the blooms would result in ammonia toxicity to salmonids under present ammonia loads.

A mathematical model developed by the USGS was used to evaluate physical effects of the proposed mid-channel dredging alternative. The model results indicate that wedge residence time would increase by 14% and that the wedge toe location would be unaffected. Increased low-flow conditions in the Green River will only slightly alter wedge characteristics.

Since oxygen demand is primarily benthic, a dredging project which only increases wedge depth (area constant) will have minimal effect on wedge D.O. concentrations because the longer residence (reaction) time will be offset by the increased dilution of SOD. The alternatives considered will have negligible impact on phytoplankton production. For the mid-sized alternative, wedge D.O. concentrations are expected to change by less than 0.1 mg/l. The proposed dredging project is not expected to enhance problems associated with ammonia toxicity and toxin spills in the estuary.

Additional studies of biological and chemical processes within the Duwamish estuary would refine assumptions and parameter values pertinent to the analysis of dredging impacts. Specifically, additional studies of spatial and temporal variations of SOD, D.O. uptake throughout the salt-wedge (requiring the reinstallation and maintenance of the Spokane St. monitor), and toxicity of RTP effluent using algal bioassays would aid in analyzing impacts of dredging in the Duwamish.

LIST OF FIGURES

LIST OF TABLES

NOTATIONS

ACE =	Army Corps of Engineers
BOD =	Biochemical Oxygen Demand
C =	Carbon
^{14}C =	Radioactive Carbon
cfs =	Cubic Feet Per Second
Chl*a* =	Chlorophyll *a*
Cl_2 =	Chlorine Concentration
CO_2 =	Carbon Dioxide
CSO =	Combined Sewer Oveflow
d =	Depth of Water Column
D.O. =	Dissolved Oxygen
DOC =	Dissolved Organic Carbon
DOE =	Department of Ecology
in sito =	Experiment or measurement done in the field
in vitro =	Experiment or measurement done in the laboratory
Km =	Kilometer = .621 miles
m =	Meters = 3.2808 ft.
m^3 =	Cubic meters = 35.28 cubic feet = 1.31 cubic yard
mgd =	Million gallons per day =0.646 cfs
mg/l =	Milligrams per liter
mil =	Million
μg/l	Micrograms per liter

MLLW = Mean Lower Low Water

N = Nitrogen

ng/l = Nanograms Per Liter

NH_4 = Unionized Ammonia

NH_3 = Ammonia

pH = Logarithm of the hydrogen ion concentration

P:R = Production to Respiration

Q_f = Freshwater inflow to the estuary

RTP = Renton Treatment Plant

SCS = S[illegible] Conservation Service

SOD = Sediment Oxygen Demand

SS = Suspended Sediment

Sta. = Station

std. dev. = Standard Deviation

$TRCl_2$ = Total Residual Chlorine

USGS = United States Geological Survey

yd^3 = Cubic Yard

ACKNOWLEDGEMENTS

Harper-Owes wishes to acknowledge the following agencies and individuals for the participation in the project:

Dr. Eugene B. Welch of the University of Washington; Municipality of Metropolitan Seattle staff including Robert I. Matsuda, James Buckley, Janet Condon, and Lucy Woo; and Dr. E. A. Prych of the U. S. Geological Survey.

INTRODUCTION

The Seattle District of the U. S. Army Corps of Engineers has undertaken studies to address the impacts of a proposed project to widen and deepen the navigation channel in the Duwamish River estuary. Several key issues were identified for the evaluation of project impacts on aquatic resources in the area. A major concern is the impact of a wider and deeper channel on water quality parameters and the dynamics of the salt wedge in the estuary. This report addresses this concern.

Existing conditions in the Duwamish estuary are documented in the following sections. Principal water quality parameters evaluated include dissolved oxygen and phytoplankton. Analysis of dissolved oxygen consumption in the saltwater wedge is emphasized. A section has also been prepared discussing sediment transport in the estuary.

Three alternative sizes are being considered for the proposed project. Impacts from the mid-sized channel are addressed on circulation, dissolved oxygen concentrations and phytoplankton in the estuary.

EXISTING CONDITIONS

SUMMARY OF MAJOR DATA SOURCES AND LITERATURE

Santos and Stoner (1972) have compiled an excellent review of the considerable quantity of data generated by the U.S. Geological Survey (USGS) between 1963 and 1967 on the Duwamish Estuary system. They have also summarized historical changes in several water quality parameters dating back as far as 1948, and the reader is referred to this publication for a review of pre-1963 information. In addition to the Santos and Stoner report, 12 other documents arising from the cooperative USGS-Municipality of Metropolitan Seattle (Metro) research effort were reviewed for this analysis and can be grouped into the following categories: 1) physical-chemical descriptions of the estuary (Isaac et.al., 1964; Stoner, 1972; Dawson and Tilley, 1972; and Stoner, 1972); 2) periphyton and phytoplankton investigations (Welch, 1969; Tilley and Dawson, 1971; Welch et.al., 1972; and Tilley and Haushild, 1975); 3) numerical models of the saltwedge portion of the estuary (Haushild and Stoner, 1973; Fisher, 1975; Stoner et.al., 1975; and Prych et.al., 1976). In addition to the above documents, pieces of unpublished USGS data pertinent to this analysis were also reviewed.

Since 1967, Metro has operated a water quality monitoring program on both the Green River and Duwamish Estuary in addition to their routine analyses of Renton Treatment Plant (RTP) effluent. The Duwamish monitoring program principally consists of an automatic monitor network at Renton Junction, East Marginal Way, 16th Ave. So. (surface and bottom) and Spokane St. (surface and bottom) (Figure 1). The monitors record temperature, salinity (conductivity), dissolved oxygen and pH (surface only) at hourly intervals. Monitor stations are typically checked and calibrated weekly. In addition to the monitor program, Metro has sampled several stations throughout the estuary biweekly from June to September.

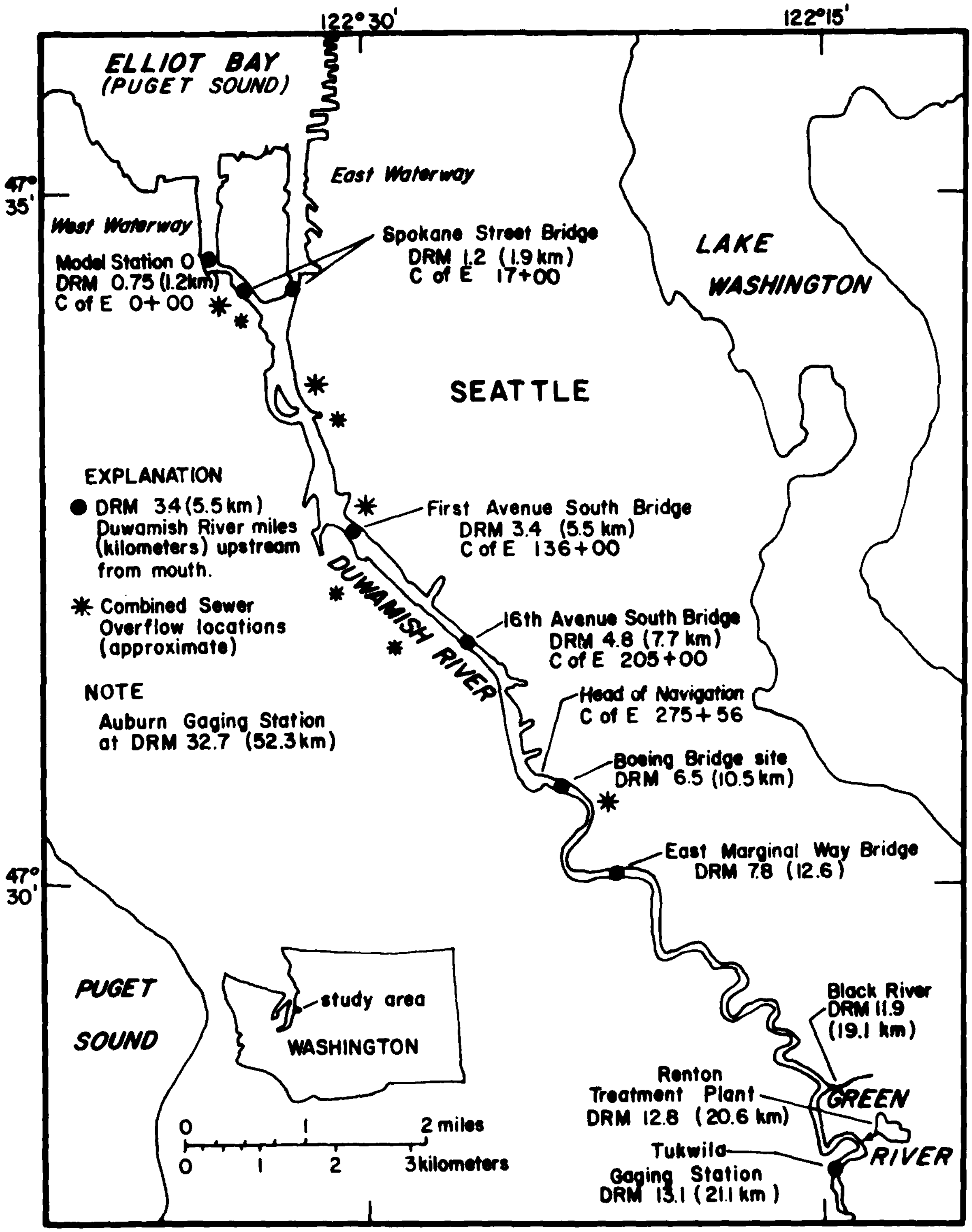


FIGURE 1

Major Features of the Duwamish River

Parameters analyzed include nitrate, ammonia, Kjeldahl nitrogen, soluble reactive phosphorus, total phosphorus, total residual chlorine and fecal coliforms. Several special studies have also been undertaken on the Duwamish. Although the quantity of Duwamish data stored in Metro files is immense, little of this information has ever been analyzed or published. Some of this information is presented here.

The Washington State Department of Ecology (DOE) has undertaken several special studies of the Duwamish estuary and the impacts of the RTP. The information reviewed for this report came from Bernhardt (1980) and Yake (1980).

Research at the University of Washington (UW) Department of Civil Engineering on ammonia dynamics within the estuary has been ongoing since summer, 1979. The results of the 1979 studies are presented in Welch and Trial (1979).

Research undertaken at the UW Department of Oceanography on distributions and bioaccumulation of polychlorinated biphenyls (PCBs) in the Duwamish Estuary, Elliott Bay and Puget Sound (main basin) was conducted from 1972 to 1977. Several publications have been released, but the study results are summarized in Pavlou and Dexter (1979).

The National Oceanographic and Atmospheric Administration (NOAA) has been conducting a study of flocculation in the estuary since 1979, and the results of the first (summer 1979) experiments were reviewed for this analysis (Feely, written communication 1980).

The U.S. Army Corps of Engineers (ACE) has undertaken numerous investigations of impacts of the dredging process in addition to periodic measurements and analyses of sediment conditions. Basic physical and chemical sediment data were reviewed.

Although the above sources contain information most pertinent to the analysis of dredging impacts, numerous other sources of data and/or reports on the Duwamish estuary exist. Studies conducted by the Port of Seattle (1978), USGS (hydrologic information) and STR (1972, 1973) have been reviewed but do not bear substantially on the outcome of this analysis.

HYDROLOGY OF DUWAMISH ESTUARY

The lower reaches of the Duwamish River estuary (approximately downstream from the head of navigation) is comprised of a salt-water wedge overlain by a fresher-water layer. Stratification is strong during high freshwater inflows which also push the wedge downstream relative to conditions during lower flows. Net circulation in the wedge is upstream due to entrainment of saltwater from the wedge to the overlying fresher-water. Net movement of the fresher-water is downstream. Circulation in the estuary is principally controlled by freshwater inflow rate, tidal action and estuary morphology.

Freshwater Inflows

Freshwater is principally supplied to the Duwamish estuary by the Green River. Flows in the Green River average 43.9 m^3/sec (1549 cubic feet per second) at Tukwila with peak flows exceeding 343 m^3/sec (12,000 cfs) and minimum flows as low as 5.5 m^3/ sec (195 cfs). Peak flows occur during winter months while low flows occur during August and September. Flows have been regulated by Howard A. Hanson Reservoir since 1961 for flood control and during summer months to augment natural river flow.

Other sources of freshwater include the Metro wastewater treatment plant located at Renton, the Soil Conservation Service (SCS) pump station and combined sewer overflows. The Renton treatment plant (RTP) presently discharges 1.6 m^3/sec (36 mgd) at river kilometer 20.6 with flows projected to increase to nearly 3.2 m^3/sec (72 mgd) by year 2000. Metro is evaluating potential adverse impacts from the RTP discharge and is considering construction of an outfall line to Puget Sound for effluent disposal. A draft EIS was submitted in January 1981 evaluating impacts.

The SCS pump station was constructed to pump drainage from the east side of the

lower Green River valley to the Duwamish River via the Black River channel. Peak flows (100-year recurrence interval) generated in the East Side Watershed have been estimated by the SCS to exceed 87 m^3/sec (3070 cfs). These flows would apparently occur when the watershed has been fully developed and drainage channels have been constructed. A project has been undertaken to construct channels from the pump station to the boundaries of the City of Kent.

Combined sewer overflows (CSO) discharge raw wastewater and storm runoff to the Duwamish estuary during periods of intense precipitation. Metro has nine flow regulator stations and the City of Seattle has three combined sewer overflows which may discharge directly to the Duwamish estuary. In addition, three combined sewer overflows are located on Longfellow Creek which flows into the West Waterway. The direct discharges to the Duwamish estuary are estimated to be approximately 1 million cubic meters per year (256 million gallons per year) while CSO discharges to Longfellow Creek amount to 60 thousand cubic meters per year. The City of Seattle has undertaken a program to control the CSO discharges to Longfellow Creek. Discharges of CSO's to the Duwamish estuary are likely to continue in the future as control programs are low in priority for both Metro and the City of Seattle.

Tides

Duwamish tides are controlled by Puget Sound tides and river stage at the head end of the Duwamish River. Tidal effects have been observed throughout the entire stretch of the Duwamish River. Changes in river stage have been recorded at Renton Junction 21 km (13 miles) upstream from the river mouth. The upstream extent of flow reversal is between river km 12.6 and km 21.0 (7.9 mi.and 13 mi.) (Dawson and Tilley, 1972).

The datum for tides in the Duwamish estuary is mean lower low water (MLLW) at a station located at river km 6.9 (mi. 4.3). Mean tide stage is 2m (6.6 feet) above MLLW. Maximum and minimum recorded stages are 4.5m (14.7 feet) above MLLW and 1.4m (4.6 feet) below MLLW, respectively. Tidal prism thickness (the sum of daily high tides minus low tides) ranges from 3 to 6 m (10 to 20 feet).

Circulation

Circulation of water within the Duwamish estuary consists of net upstream movement in the saltwater wedge and net downstream movement in the fresher water overlying the wedge. The wedge moves upstream and downstream as a function of tide stage and freshwater inflow. During periods of low freshwater inflows and high tides, the wedge has extended as far as km 16 (mi. 10) (Stoner, 1972). At freshwater inflows greater than 28 m^3/sec (1000 cfs) the wedge remains downstream of the East Marginal Way bridge (km 12.6 or mi. 7.9) regardless of tidal stage (Stoner,1967).

The net upstream movement of water within the wedge is caused by entrainment of saltwater from the wedge to the overlying fresher-water. Entrainment velocities were found to vary with freshwater inflow and tidal prism thickness (Stoner, 1972). An equation relating entrainment velocity to freshwater inflow and tidal prism thickness developed for the USGS model (Prych et al, 1976) predicts entrainment velocities similar to the velocities computed by Stoner.

Estuary Geometry

Physical characteristics of the estuary are summarized in Table 1. Total volume of the channel exceeds 20 million cubic meters below MLLW. Deepest reaches in the East and West Waterways exceed 16 m (50 feet) while shallowest reaches near the head of navigation are close to 3 m (10 feet) in depth.

TABLE 1

Approximate Existing Physical Characteristics of

The Lower Duwamish Estuary (Relative to MLLW)

Reach	Volume (mil m^3)	Volume (mil yd^3)	Average Channel Depth (m)	Average Channel Depth (ft)	Average Surface Width (m)	Average Surface Width (ft)
East Waterway	6.4	8.4	15	50	230	750
West Waterway	5.4	7.0	15	50	230	750
Duwamish Waterway	8.6	11.2	10	33	160	525

EXISTING WATER QUALITY CONDITIONS

Dissolved Oxygen

A considerable number of studies have been undertaken on the Duwamish which have focused on dissolved oxygen (D.O.) conditions and dynamics within the estuary. Because the residence time of the estuary wedge may increase and D.O. decrease as a result of dredging operations (Haushild and Stoner, 1973), D.O. has been a key parameter in this analysis. The D.O. is especially significant in the Duwamish River system, which supports commercially and recreationally important runs of steelhead and cutthroat trout and chinook, coho, and chum salmon.

Dissolved oxygen conditions within the estuary have changed markedly over the years primarily in response to changes in wastewater discharge characteristics. Historically, the lowest D.O. concentrations have always been observed in the water at the toe of the salt wedge (16th Avenue South), primarily because of the oxygen demand exerted on this water mass as it moves landward up the estuary without reaeration (Santos and Stoner, 1972). Minimum D.O. concentrations at this station have always occurred during the months of August, September and October. During 1949 and 1956, mean concentrations for these months at km 7.7 (mi 5.6) were 4.8 and 5.9 mg/l, respectively. But by 1967 the mean D.O. for the same period had dropped to 2.9 mg/l, with periodic sags to 0.9 mg/l (Santos and Stoner, 1972). Detailed comparisons between early and recent years, however, are complicated by the differences in sampling procedures used (e.g. monthly, weekly or hourly samples). Following the interception of the majority of untreated and partially treated discharges into the Duwamish by 1970, oxygen conditions within the estuary improved markedly. An analysis of 1976-1980 Metro monitor data reveals that D.O.

concentrations at Sta. 7.7 from August to October averaged 5.3 mg/l, and minimum concentrations always exceeded 3.5 mg/l. It appears likely from this preliminary review that D.O. levels in the wedge may have returned to early 1950 levels. It should be emphasized that freshwater discharge from the Green River during all of the periods mentioned above (i.e. Aug.-Oct. 1949, 1956, 1967, 1976-1980) were all similarly low, and therefore residence time of the wedge should be equivalent. A more detailed discussion of the factors influencing wedge dissolved oxygen occurs in the following section.

Although dissolved oxygen conditions in the wedge do not appear to have changed significantly since 1970, surface water D.O. content at the freshwater end of the estuary has exhibited periodic sags in recent years. Indeed it is not uncommon in recent years for surface D.O. concentration to be at or below bottom values. The D.O. sag study reported by Yake (1980) observed a sag of 2.8 mg/l between Renton Junction and km 7.7 during fall, 1979. Yake attributed the bulk of oxygen consumption within the river to nitrification of the very large quantities of ammonia discharged from the RTP facility. Between 1972-73 and 1979-80, summer ammonia concentrations at km 7.7 surface have increased over 5-fold, solely as a result of increased effluent discharges from RTP (Figure 2). It appears likely, therefore, that nitrification of the RTP effluent is the cause of surface D.O. depletions.

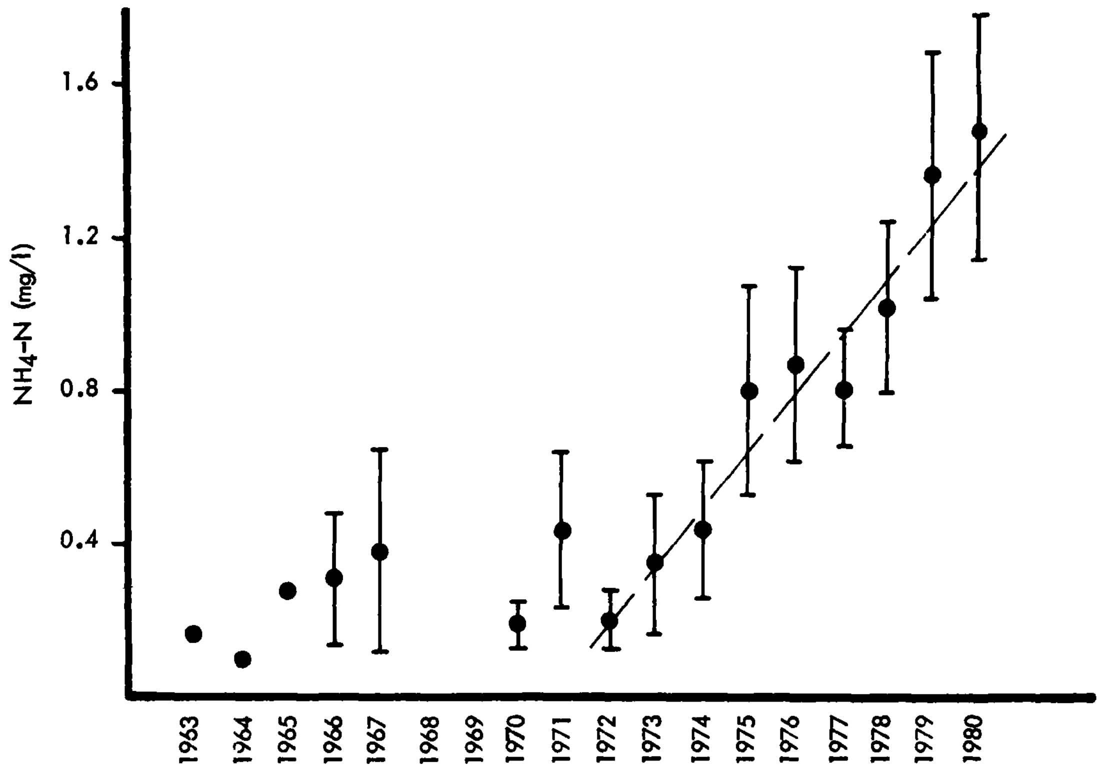


FIGURE 2

Mean (± 1 std. dev.) ammonia concentration at 16th Ave. So., surface during June to September.

Oxygen Consumption Within the Wedge

Early versions of the USGS model treated oxygen consumption in the wedge as a function solely of suspended and dissolved BOD. Later modifications of the model divided total oxygen consumption in the wedge into suspended-dissolved (75%) and benthic (25%) components (Stoner et. al., 1975). The magnitude of each component was determined by comparison of total wedge consumption (derived from a best-fit of D.O. monitor data and model parameters) during summer 1970 and 20 wedge BOD samples taken at low flow during August 1969 - March 1970 (mean BOD_5 = 1.4 mg/l). The latest modification of wedge oxygen consumption rates in the USGS model increased all rates by 25% to compensate for an increase in entrainment velocity above that used in previous runs (Prych et.al., 1976). In this last version all oxygen consumption was assumed to occur in the suspended-dissolved component.

The question of suspended-dissolved vs. benthic oxygen consumption in the Duwamish estuary is insignificant for the purposes of modeling dissolved oxygen concentrations within the existing wedge. In either case, the tidal-averaged wedge dissolved oxygen concentrations at km. 7.7 would be identical as long as the same monitor information was used to calibrate model parameters. However, with respect to the proposed dredging project, predictions of changes in the dissolved oxygen regime are dependent upon whether oxygen consumption occurs primarily in the wedge column or sediments (or both). For example, if oxygen consumption occurs wholly within the wedge column and if this consumption rate (BOD) is constant, then minimum wedge dissolved oxygen concentrations near the wedge toe would simply be inversely proportional to residence time. This is essentially the rationale used in Haushild and Stoner's (1973) analysis. However, if the bulk of respiration occurs within the sediments, then an increase in wedge depth, if

accompanied by an equivalent increase in detention time, will result in dissolved oxygen conditions identical to those existing before deepening. That is, the increased residence time of water within the wedge exposed to a given sediment oxygen demand (SOD) is offset by the greater volume of water in which the SOD exerts. If the areal-weighted SOD and net wedge discharge (i.e. entrainment) are equivalent before and after dredging, then the before and after D.O. regimes will be exactly the same. This perhaps unlikely relation can also be demonstrated mathematically by performing an oxygen mass balance on an idealized (well-mixed) wedge segment (Figure 3).

$$Q_1 * C_1 + Q_3 * C_3 - Q_2 * C_2 - Q_4 * C_4 = SOD * A$$

where Q = discharge (m^3/day)

C = dissolved oxygen concentration (gms/m^3)

SOD = sediment oxygen demand (g/m^2/day)

A = bottom area (m^2)

Subscript 1 refers to water moving into the idealized wedge segment from the adjacent downstream segment;

Subscript 2 refers to water moving out of the wedge segment to the adjacent upstream segment;

Subscript 3 refers to water mixed downward into the wedge segment from the overlying fresher-water;

Subscript 4 refers to wedge water mixed upward into the overlying fresher-water.

The above equation simply illustrates that if discharges, which are principally a function of entrainment, and inflow concentrations remain constant then an increase in wedge depth will have no effect on wedge D.O. concentrations (C_2 and C_3). Volume

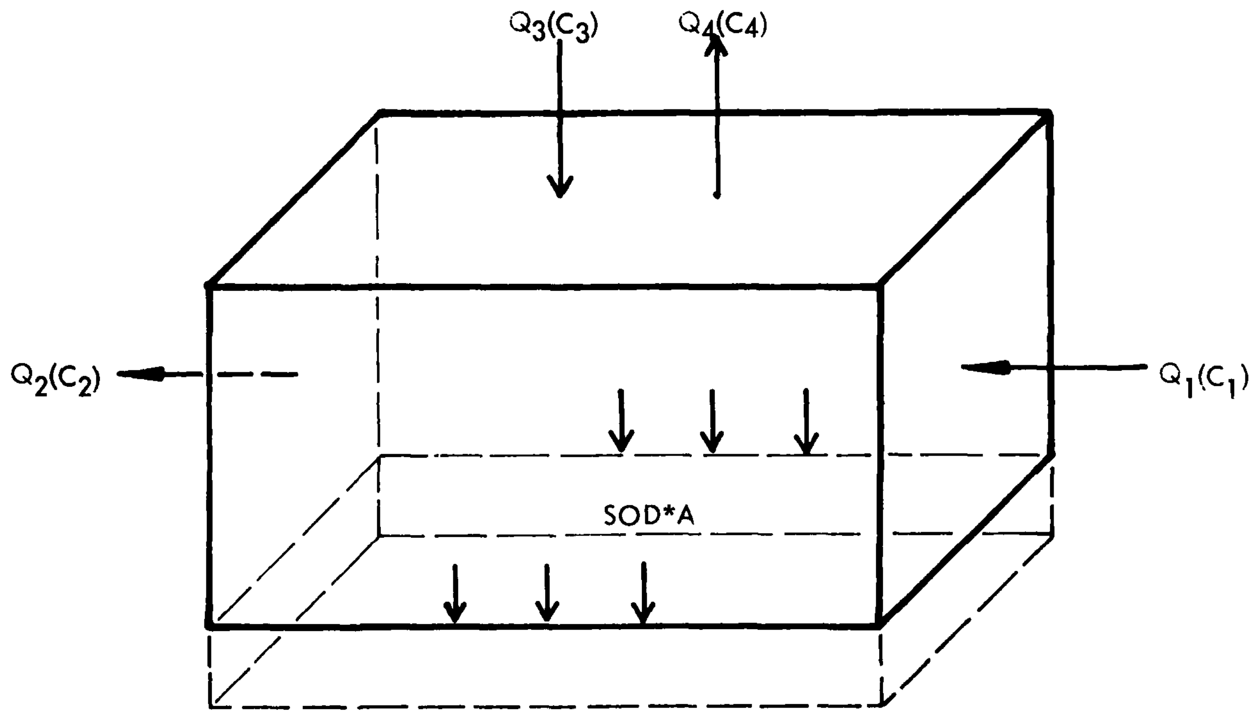


FIGURE 3

Idealized wedge segment showing principal oxygen fluxes assuming biochemical oxidation is a solely benthic process.

and residence time, per se, do not influence wedge D.O. concentrations.

As discussed above, Stoner et. al. (1975) estimated that 75% of the total wedge oxygen consumption was due to suspended-dissolved components and 25% to sediment components. Using the revised entrainment velocities of Prych et.al. (1976) and the same BOD data used by Stoner et.al. (1975), the oxygen demand of the suspended-dissolved fraction is calculated at 60% and the sediments is 40%. However, given the problems in extrapolating low-level BOD_5 test results to in situ respiration rates, the fact that many of the BOD_5 tests were performed prior to removal of Diagonal Way primary effluent discharge and subsequently compared with post-Diagonal Way wedge oxygen consumption rates, and the sensitivity of the water column vs. sediment calculations to assumed entrainment velocities, the above values of relative oxygen consumption must be considered very rough approximations.

The most detailed survey of sediment oxygen demand in the Duwamish Estuary was undertaken by Metro primarily during August, 1973 when 41 separate determinations of in situ SOD rates were performed throughout the estuary (Figure 4) (R.I. Matsuda, written communication, 1973). Comparison of areal-weighted SOD rates determined from this survey with total wedge O_2 consumption calculated with the use of Metro monitor data at Spokane Street and 16th Avenue South during the same periods would therefore give a more reliable estimate of the relative contributions of suspended-dissolved and benthic oxygen consumption rates in the estuary. Rather than using the USGS model to calculate total consumption, a more direct approach was undertaken using daily average salinities and dissolved oxygen concentrations at the Metro monitor stations. As illustrated in Figure 3, if oxygen consumption is assumed to be entirely benthic, then a determination

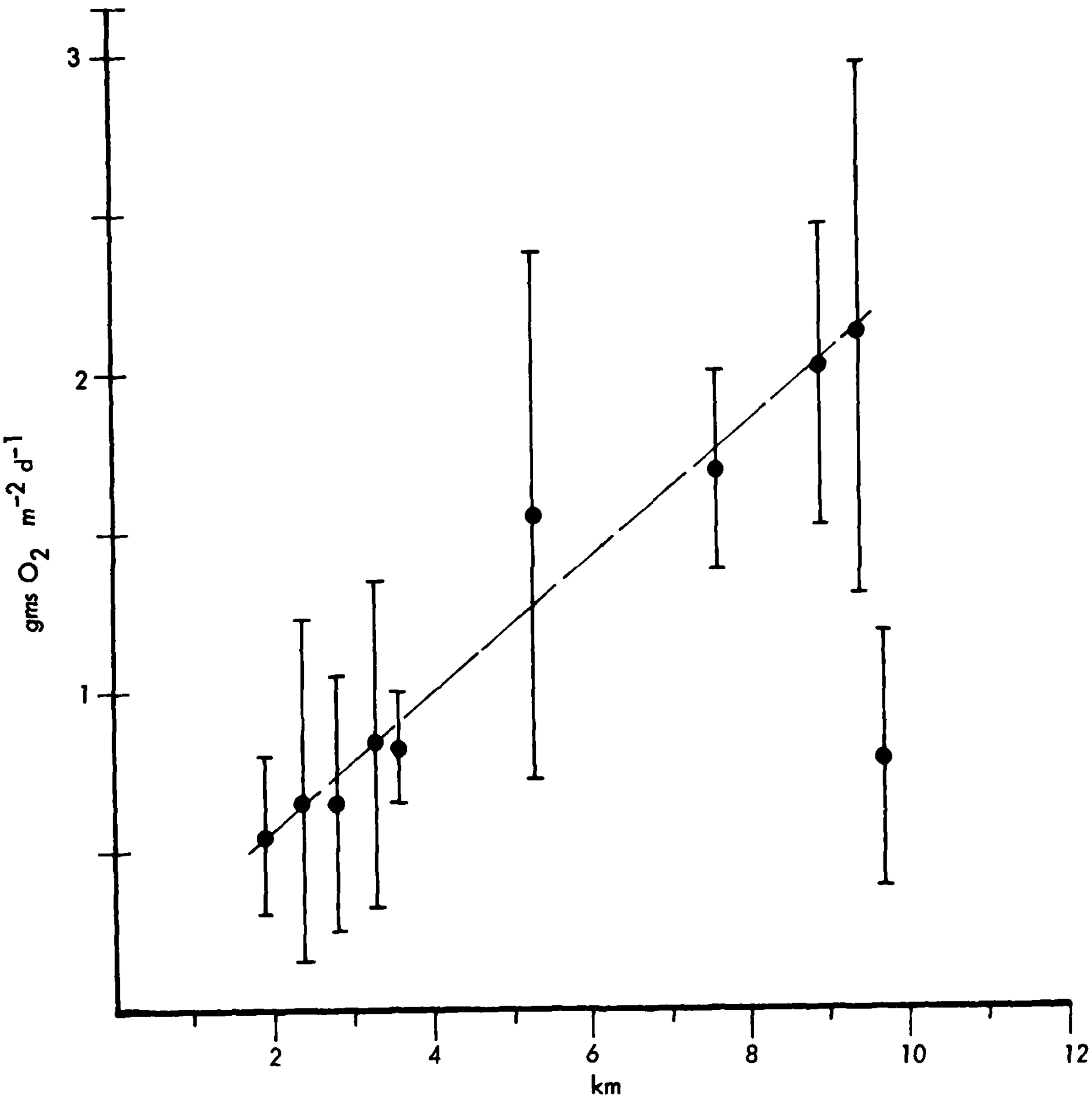


FIGURE 4

Sediment oxygen demand as a function of longitudinal distance from the estuary mouth during August, 1973. (mean $\pm$ 1 std. dev.) (R. I . Matsuda, written comm. 1975).

of flux rates of oxygen to and from the wedge would yield an estimate of SOD. Comparison of this "total wedge" SOD with measured SOD for August, 1973 would therefore give an estimate of the non-sediment contribution.

From 1967 to 1976 Metro has recorded hourly values of temperature, salinity and dissolved oxygen from automatic monitors located 0.9 m below water surface and 0.9 m above channel bottom at 16th Avenue South and Spokane Street. Surface monitors are also located at Renton Junction (above effluent discharge) and East Marginal Way (Figure 1). The positions of the deeper two monitoring stations provide estimates of concentrations of wedge water moving landward and mixed (fresh and salt) water moving seaward at two points in this salt-wedge estuary (Figure 5). Salinity and velocity data taken during several surveys in 1967 and 1968 reinforce the assumption that the Metro surface and bottom monitors are sampling water representative of the bulk of water moving in a net seaward and landward direction, respectively (Stoner, 1972; Stoner et.al., 1975). If this assumption is correct, then discharges in various regions of the estuary can be evaluated by simultaneously solving flow and mass balance equations for salt in the estuary. That is, knowing tidal-averaged values of salinity at four points in the estuary, and incoming freshwater discharge and salinity concentration, the flow and mass balance equations given below can be solved to yield discharges at six areas in the estuary. Refer to Figure 5.

Flow Balance Equations:

$$Q_0 + Q_2 = Q_3$$

$$Q_5 + Q_2 - Q_6 = Q_1$$

$$Q_3 + Q_5 - Q_6 = Q_4$$

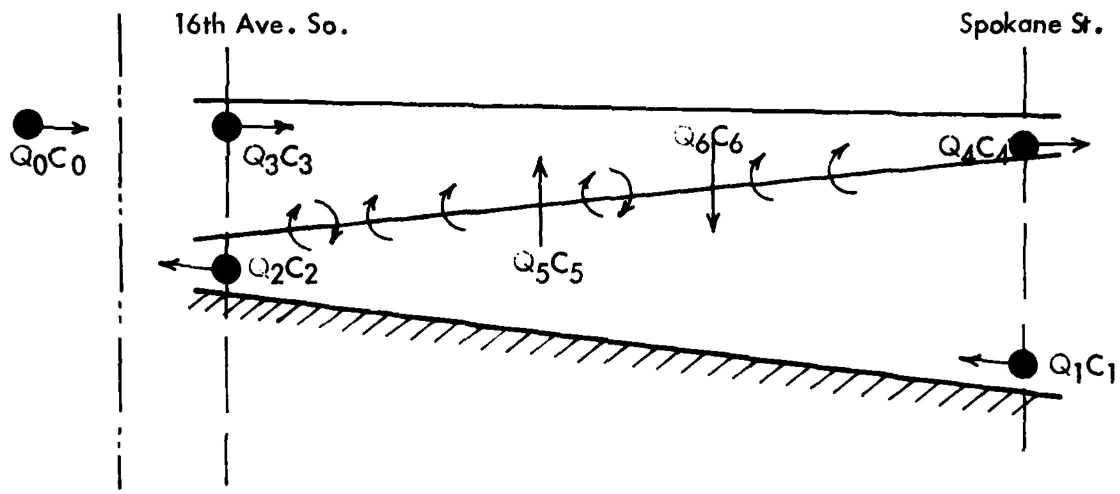


FIGURE 5

Idealized longitudinal section through the salt wedge portion of the Duwamish Estuary, showing net two-layer flow and Metro monitor stations.

Mass Balance Equations:

$$Q_0 C_0 + Q_2 C_2 = Q_3 C_3$$

$$Q_1 C_1 + Q_6 C_6 - Q_5 C_5 = Q_2 C_2$$

$$Q_3 C_3 + Q_5 C_5 - Q_6 C_6 = Q_4 C_4$$

Knowns:

All C's, Q_0

Assumptions:

$C_5 = C_1 + C_2/2$

$C_6 = C_3 + C_4/2$

Arithmetic Average C = Tidal Average C

C's are representative of bulk of surface and bottom net flow

Where:

Subscript 0 refers to the mixture of Green River water and RTP effluent;

Subscript 1 refers to water in wedge at Spokane St.

Subscript 2 refers to water in wedge at 16th Ave. So.

Subscript 3 refers to surface water at 16th Ave. So.

Subscript 4 refers to surface water at Spokane St.

Subscript 5 refers to wedge water mixed into overlying fresher-water

Subscript 6 refers to overlying fresher-water mixed into wedge.

Of the four assumptions listed above, the last is probably the most likely to be violated, but only at Spokane Street surface, where during summer low flows the estuary grades into a partially mixed type showing more of a sharp gradation of salinity over the top two meters. However, even this monitor station probably succeeds in obtaining a reasonable approximation of the seaward flow, since most of the flow does occur fairly near the surface.

The mass balance model therefore provides estimates of flows through and between the two water layers at 16th Avenue South and Spokane Street. It should be mentioned that the model described here calculates entrainment of surface water down into the wedge as well as from the wedge upward, since the salinity at 16th Avenue South, bottom is typically slightly lower than Spokane Street bottom. The magnitude of this downward entrainment discharge was typically 1/10 the upward entrainment flow.

Knowing flows at each of the five locations designated in Figure 5, the flows were then multiplied by average daily concentrations (D.O.) to get oxygen fluxes. These fluxes were then used to compute the total wedge consumption occurring between km 1.9 and km 7.7, expressed as an SOD. The total area (MLLW) between the two bridges was determined from NOAA charts at 1.115×10^6 m^2. The model was run by executing 10-day moving averages with a 5-day time step. Days with any piece of missing data were totally rejected from the model.

The average SOD computed for August, 1973 was 1.05 $gm/m^2/d$. The average measured SOD from km 1.9 to km 7.7, determined from Figure 4 weighted for width was 1.03 $gm/m^2/d$. While the comparison of measured to calculated SOD rates is somewhat complicated by the fact that longitudinal values of measured SOD increase

as one proceeds landward, the close agreement of the two values does indicate that the majority of oxygen consumption in the wedge takes place in or on the sediments. A value in the vicinity of 90% total wedge consumption attributable to benthic processes therefore seems justifiable.

The mass-balance model generates estimates of average SOD rates from Spokane Street to 16th Avenue South. The USGS model, using the same D.O. data at the Metro monitor stations for parameter calibration, generates an average oxygen consumption rate in the wedge water (BODw) expressed on a volumetric basis (gms m^{-3} day $^{-1}$). Both models do not incorporate changes in SOD (or BOD) with longitudinal distance in the estuary. As a check of the mass-balance model, apparent SOD rates were divided by the mean depth of the wedge (7.4m during low flows) and the resulting BODw compared to BODw values for the same time period from the USGS model. The average BODw values for June-September, 1967-68 calculated with the three model runs were:

Model	BODw (gms m^{-3} day $^{-1}$)	Source
mass-balance	0.55	This study
USGS	0.61	Stoner et.al., 1975
USGS	0.77	Prych et.al., 1976

The BOD w values calculated with the mass-balance and USGS-Stoner et.al. (1975) models are in close agreement, but the USGS-Prych et.al. (1976) model generates values of considerably greater magnitude. As stated above, the Prych et.al. (1976) version used entrainment velocities 25% greater than previous USGS runs. Given the close agreement with the Stoner et.al. (1975) model, and the variable nature of entrainment velocities,

it appears that the mass-balance model describes oxygen consumption in the wedge as adequately as the other models.

The rather large percentage of total oxygen consumption in the wedge attributable to benthic processes (about 90%) is supported by a consideration of the flocculation process in the estuary. Since the floc formed upon mixing Green River water with Elliott Bay water has been shown to be principally organic (Feely, 1980), dissolved BOD inputs entering the estuary with fresh water (upstream sources, RTP effluent and CSO's) would become associated with the particulate phase and settle in the quiescent salt-wedge reach. BOD inputs to the estuary occur principally from these fresh-water sources.

The mass-balance model was run for all monitor data from 1967 to 1976 and the results are presented in Figure 6 as total annual wedge oxygen consumption between Spokane St. and 16th Avenue So. Although oxygen consumption appears to have been relatively constant from 1970 to 1976, 1967-69 values are roughly double those of 1970-1976. The reduction in oxygen consumption from 1969-70 is coincident with interception of raw and partially treated waste discharges to the Duwamish, particularly the removal of the large Diagonal Way treatment plant discharge (primary treatment) in late 1969. Summer oxygen consumption rates have historically been substantially higher than winter rates; approximately 70% of the annual oxygen consumption occurred from May to October.

Sources of BOD Inputs

Sources of organic enrichment of the sediments include upstream sources, RTP effluent, combined sewer overflows (CSO's) and algal biomass produced and sedimented

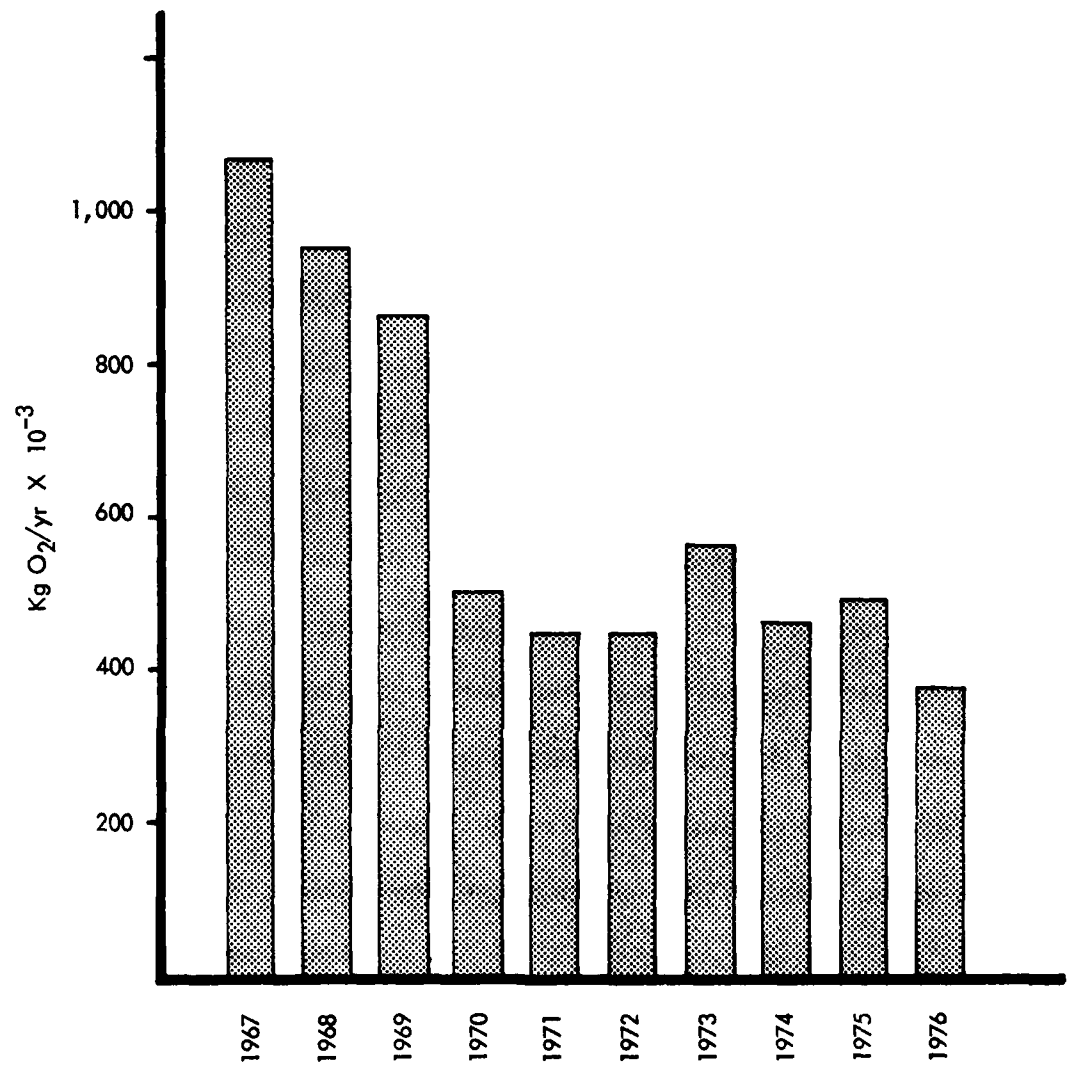


FIGURE 6

Annual Dissolved Oxygen Consumption in the wedge between Spokane St. and 16th Ave. So. calculated using the mass-balance model (see text).

in the lower estuary. Elliott Bay inputs are assumed to be negligible. No information is available on upstream BOD inputs, but they are likely to be substantial. Present carbonaceous BOD loading from RTP is 580 x 10^3 kg BOD_U/yr (RTP records, 1977-1980). Nitrification is apparently inhibited in the saltwater portions of the estuary, so nitrogenous BOD inputs were not included in this analysis (Welch and Trial, 1979). Contributions from sewer overflows discharging directly into the Duwamish are 85 x 10^3 kg BOD_U, not including the Longfellow Creek (km 1.2) loads (305x10^3 kg BOD_U/yr) (Metropolitan Engineers, 1979). The impact of the Longfellow Creek load on Duwamish D.O. concentrations is difficult to assess since much of the input may leave the estuary during ebb tide conditions. In addition, no local elevation in SOD rates was found during the August, 1973 survey (Figure 4). Since much of the CSO (and upstream) input occurs during the winter-spring months, the load from this relatively large source may not be reflected during summer conditions. A more detailed investigation of SOD rates throughout the year would be required to assess the impacts of localized discharges.

Depressions in wedge D.O. following algal blooms have been reported by Welch (1969). In order to assess the magnitude of this D.O. depletion, total algal production figures must be known. Correlations were established between the difference between daily maximum and minimum dissolved oxygen concentrations at 16th Avenue So. and Spokane Street surface monitors and daily gross production measurements made during 1966 (Welch, 1969). Estimates of production using light and dark bottles agreed well with estimates using the free-water change method of Odum (1956). Since algae typically respire a considerable amount of daily production (gross) shortly after photosynthesis, these gross production estimates must be converted to net values. Algal production to respiration (P:R) ratios

commonly observed in enriched estuary systems average roughly 1.2:1. Total community P:R values observed in the Duwamish commonly are less than unity, even during bloom periods, and suggest that a P:R ratio for algae of 1.2:1 is appropriate. ^{14}C uptake data was not used in this analysis because the method gives results intermediate between gross and net production. Conversion of daily maximum-minimum D.O. concentrations in the Duwamish to potential BOD production was done for the years 1964 to 1980 and the results are presented in Figure 7. Comparison of these values to the above estimates of CSO and RTP loads reveals that on an annual basis phytoplankton is not a major BOD source (Figure 7). Periphytic and/or planktonic algae washed into the estuary from upstream (about 2 µg/l chl a) also do not constitute an important BOD source (28×10^3 kg BOD U/yr.).

Although phytoplankton do not appear to be important sources annually, during and immediately following a bloom they may exert a demand sufficient to cause temporary reductions in wedge D.O. Comparison of maximum-minimum surface D.O. data to calculated SOD rates in 1972 reveals an apparent correlation between blooms and oxygen consumption rates in the wedge (Figure 8). The relation is somewhat obscured by the moving averages used to compute SOD rates. Seasonal variations in SOD rates during a one-bloom year (1972) contrast to the virtually unchanging condition observed during a nearly continuous bloom year (1967, Figure 9) and a no-bloom year (1976, Figure 10). Changes in bloom intensity and/or severity may therefore significantly affect periodic oxygen minima during the growing season.

Ammonia

As discussed above, the RTP is the principal source of ammonia to the estuary,

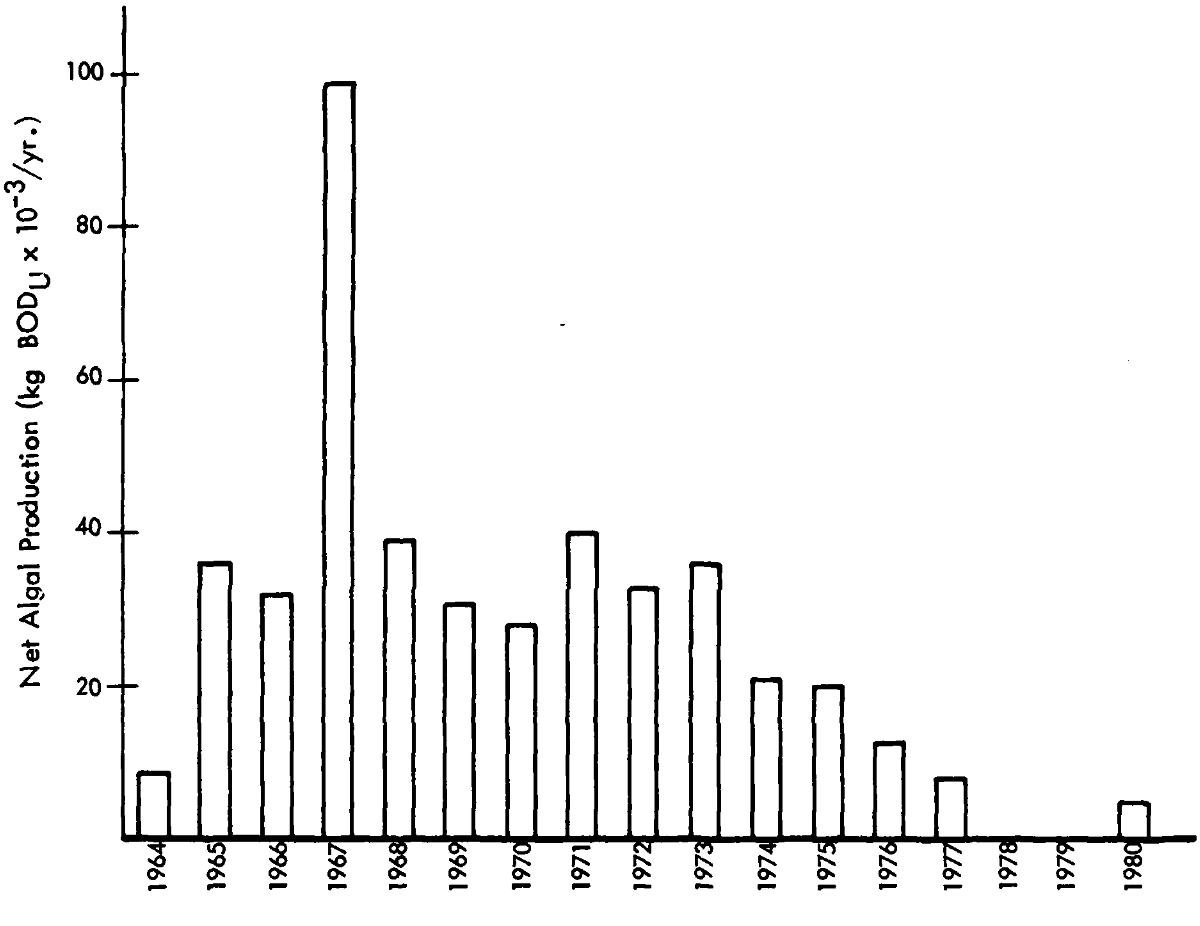


FIGURE 7

Estimated annual net algal production in the Duwamish estuary 1964-1980. (Assumptions: P:R = 1.2, Max.-Min. D.O. proportional to gross production: 1 mg/1 day = 290 kg BOD$_U$) 1964-65 data from Welch (1969).

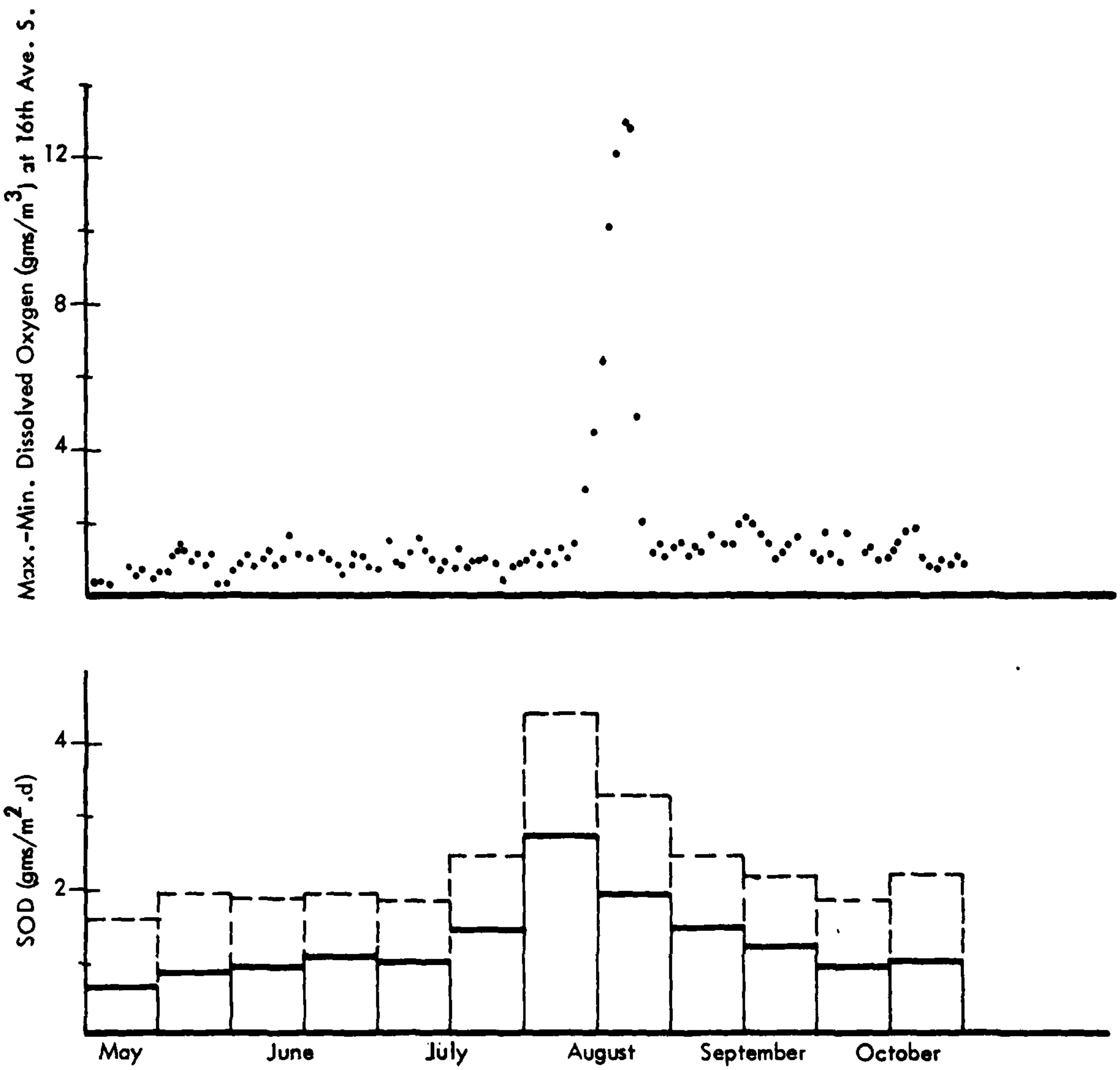


FIGURE 8

Maximum-Minimum Dissolved Oxygen and Computed SOD Rates for Summer 1972

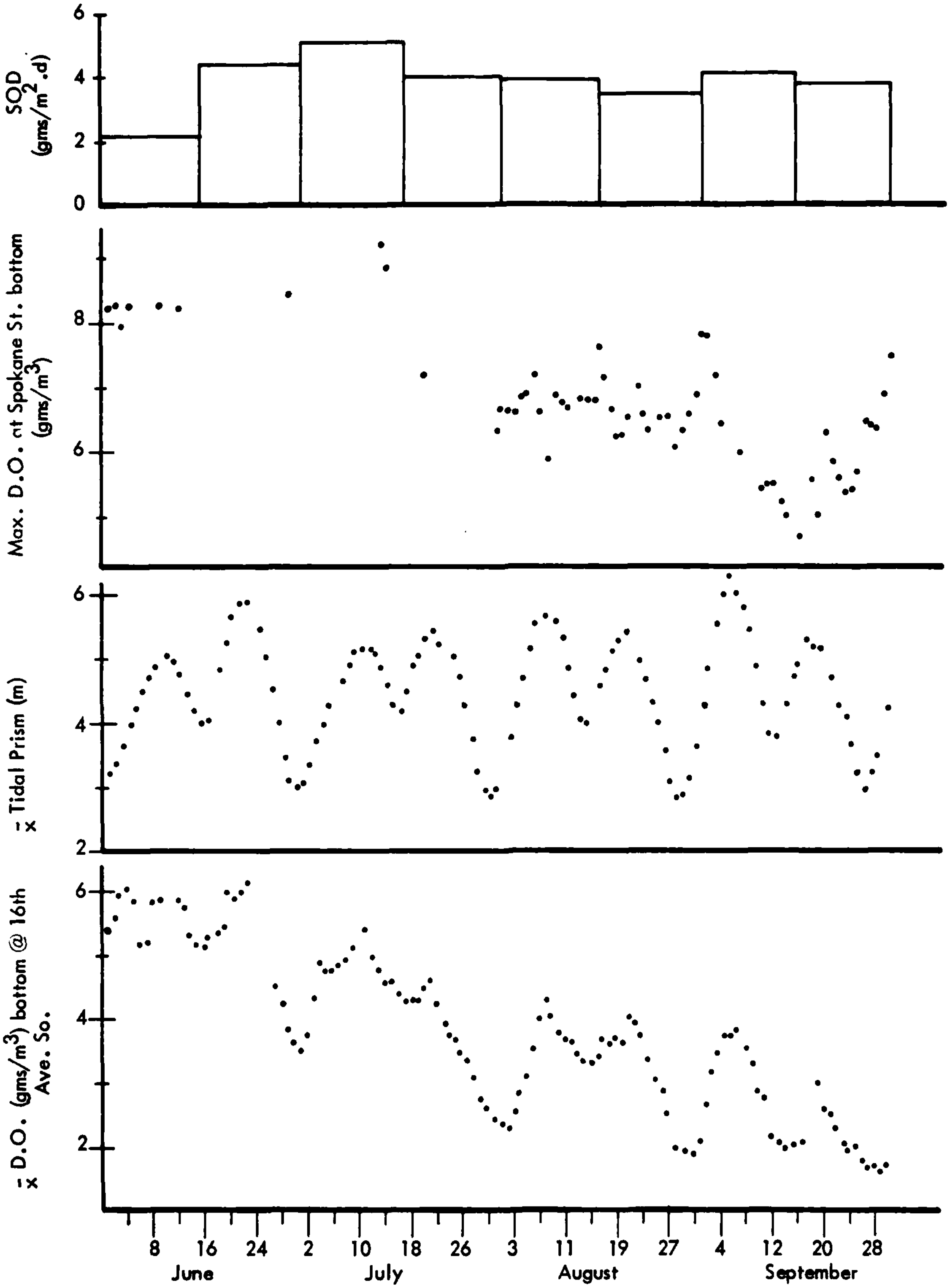


FIGURE 9

Sediment oxygen demand (SOD), salt-water inflow D.O. concentration, mean tidal prism, and mean D.O. concentration at 16th Ave. So. bottom during summer, 1967.

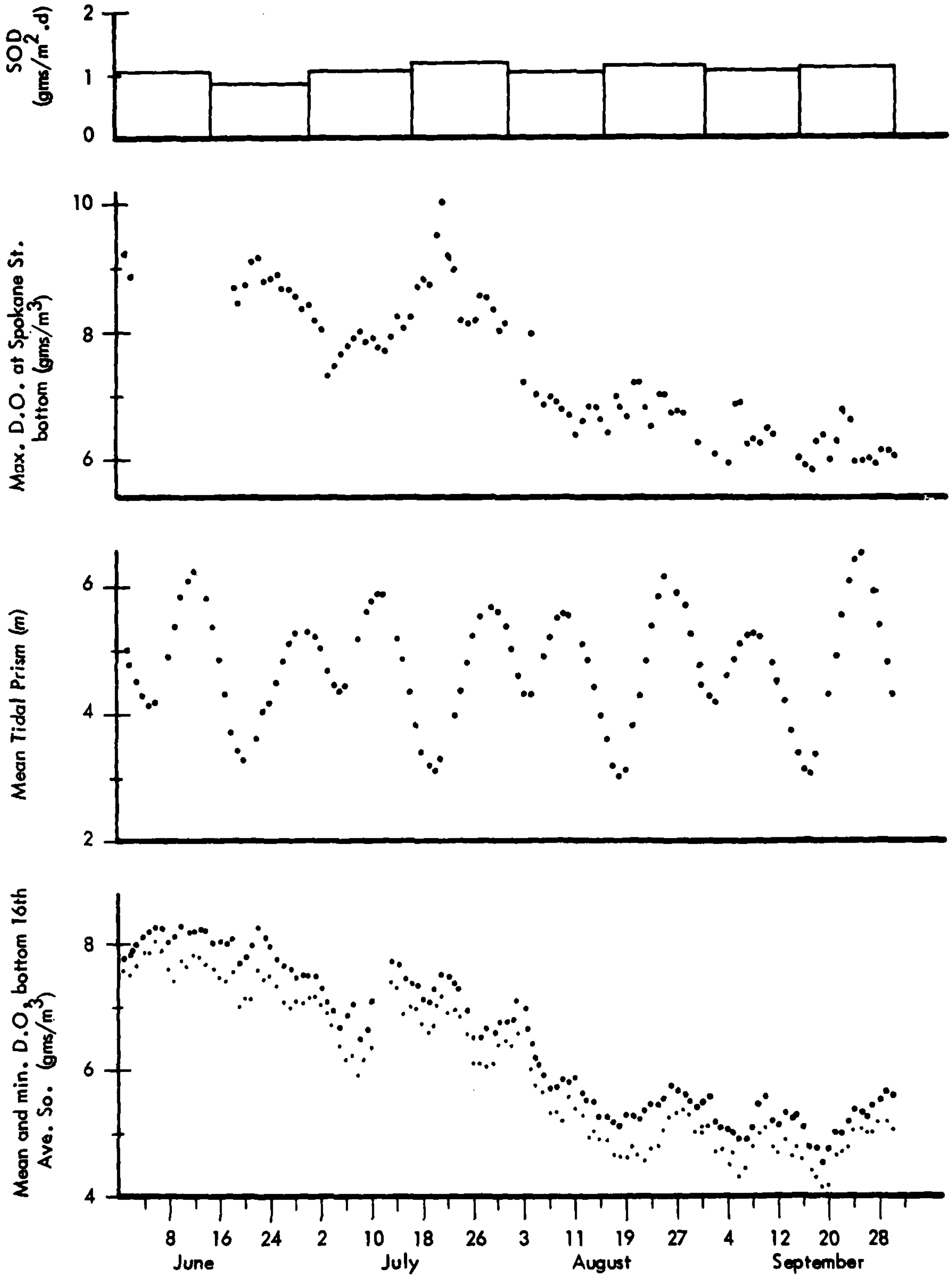


FIGURE 10

Sediment oxygen demand (SOD), salt-water inflow D.O. concentration, mean tidal prism, and mean and minimum D.O. concentration at 16th Ave. So., bottom during summer, 1976.

presently adding some 15 times more ammonia than upstream sources. However, Welch and Trial (1979) have hypothesized that sediments in the Duwamish estuary may function as an ammonia source during the summer months. Although dye and salt budget studies show that there is very little mixing of surface water into the saline wedge, concentrations of ammonia increase in the wedge water upstream from the estuary mouth. Both bottom and surface concentrations have increased at km 7.7, although wedge concentrations are presently roughly 1/2 surface values. A possible mechanism which accounts for the observed distributions of ammonia involves flocculation and/or sedimentation of organic nitrogen compounds from surface waters to the wedge or sediment, where ammonia is subsequently mineralized and released. This hypothesis is supported by Duwamish flocculation experiments which indicate that the floc formed with mixed Green River (below RTP) and Elliott Bay waters is predominantly organic (i.e. high nitrogen content) (Feely, 1980). RTP effluent is therefore implicated as the source for high ammonia concentrations in the wedge (mean 1980 NH_4-N=0.64 mg/l).

Ammonia toxicity can be a problem for salmonids when the unionized (NH_3) concentration exceeds 20 μg/l (EPA, 1973). The proportion of total ammonia which is unionized is a function of pH, temperature and salinity of the water. Summer maximum pH and temperature in the salt wedge was 8.0 and 15° C, respectively in 1979 (Welch and Trial, 1979). For these conditions, the total ammonia would have to exceed 0.5 mg/l to become toxic in the wedge. However, in the surface water at km 7.7 where algal blooms historically have elevated pH to 9.0, the possibility of an ammonia toxicity problem is far more likely. For typical bloom conditions at km 7.7 (Temp = 20° C),

the ammonia concentration necessary to achieve toxic conditions (for several hours) is only 0.07 mg/l. The situation, however, is complicated by the noticeable lack of algal blooms in the estuary in recent years.

1979-80 mean summer concentrations of ammonia at km 7.7 surface and bottom were 1.7 mg/l and 0.8 mg/l, respectively. The corresponding unionized ammonia concentrations for worst-case 1979-80 conditions were 60 μg/l for the surface (pH = 8.0, temp = 20°C) and 25 μg/l for the bottom (pH = 7.9, temp = 15°C) at km 7.7. Based on these calculations it appears that unionized ammonia may at times reach levels sufficient to stress resident or migratory populations of salmonids, especially juveniles.

Phytoplankton

Hydrographic Factors

The timing of phytoplankton blooms in the Duwamish estuary has been shown to be controlled by hydrographic factors (Welch, 1969; Welch et.al. 1972). Because the residence time of water is relatively short and turbulence normally great, phytoplankton do not reside long enough in a sufficiently favorable light climate to accumulate enough biomass in excess of losses to register a "bloom". A convenient criterion to indicate a bloom here is enough photosynthetic activity to raise oxygen above saturation and pH above normal, which is about 7.5.

Phytoplankton blooms are usually restricted to the August - September period when river discharge is lowest. The low river discharge rate results in maximum water residence time and minimum vertical turbulence. Blooms also appear to be favored by neap tide conditions (high lows and low highs) when tidal exchange and entrainment are minimal.

During these conditions (low tidal prism) residence time in the top two meters is increased and the low-salinity region of surface water extends further down the estuary. Since many of the recorded blooms at km 7.7 have been found to be due to freshwater algal species (Welch,1969; Prych et. al.,1976) which may be inhibited by salt water, neap tide conditions may increase the favorable residence time for these organisms. Little is known, however, of the salinity tolerances and origin of the bloom species.

During low flow and low tidal prism situations, the estuary changes from a stratified estuary (distinct freshwater layer overlying a saline wedge) to one that is partially mixed showing more of a gradation of salinity with depth. The partially mixed condition allows for greater stability in the surface four meters than if the estuary were stratified. This may be important to phytoplankton in terms of available light. However, light measurements taken in 1966 indicate that even during high tidal prism conditions algae in the upper mixed layer were apparently exposed to adequate light levels (greater than 8 gcal cm^{-2} hr^{-1} or 200 μeinsteins m^{-2} sec^{-1}) yet did not register a bloom (Welch 1969). Although stability would be detrimental to prolonged diatom production in water with long detention time, it is probable that stability is beneficial in the Duwamish, as it is in Puget Sound (Winter et. al.,1975).

Productivity and biomass profiles along with light attenuation estimates showed that the August, 1966 bloom resulted largely from growth in the upper one to two meters. Carbon-14 productivity measurements and chlorophyll _a_ determinations indicated that the average growth rate in the 0 - 1 m interval was about 0.6 day^{-1} and in the 0 - 4 m interval only about 0.3 day^{-1}. However, detailed cell count, chlorophyll _a_ and water

column mixing data from a bloom during August 1967 (Prych et. al.,1976) reveal that increases in concentrations in the estuary correspond to a growth rate (less sedimentation) of 2.2 day^{-1}, which is near the upper limit of reported growth rates for algae (Zison et. al.,1978). During this and other bloom periods, algal photosynthesis (per unit chlorophyll a) in the top two meters appears to have proceeded at rates comparable to those observed in nutrient and light saturated laboratory cultures (Laws and Bannister, 1980). If growth rate is rapid, then the principal controlling factors of bloom formation are upstream inoculum concentrations and residence time in a favorable (e.g. salinity and light) environment.

Figure 11 confirms that blooms actually result from the stable water column, which allows plankton to remain in the upper two meters where available light supports a growth rate high enough for production to sufficiently exceed losses. Two thirds of the water column chl a was contained in the top 2 m.

In addition, if one uses a simple model for biomass that contains growth rate, travel time and salt water dilution between East Marginal Way and 16th Avenue South (the typical observed distribution of actively growing bloom species) during low-flow and neap tide conditions, the effect of inoculum concentrations can be evaluated (Table 2). The model illustrates the importance of inoculum concentrations in producing blooms at km 7.7 and also suggests that growth rate must be extremely rapid to produce blooms. That is, any factor reducing the growth rate will result in large reductions in the bloom. It is interesting to note that bloom conditions at km 7.7 coincided with increases in algal biomass (low tide sampling) at km 12.6 during 1966 and 1967. In addition, the large and frequent blooms of 1967 (Figure 7) coincided with summer chl a

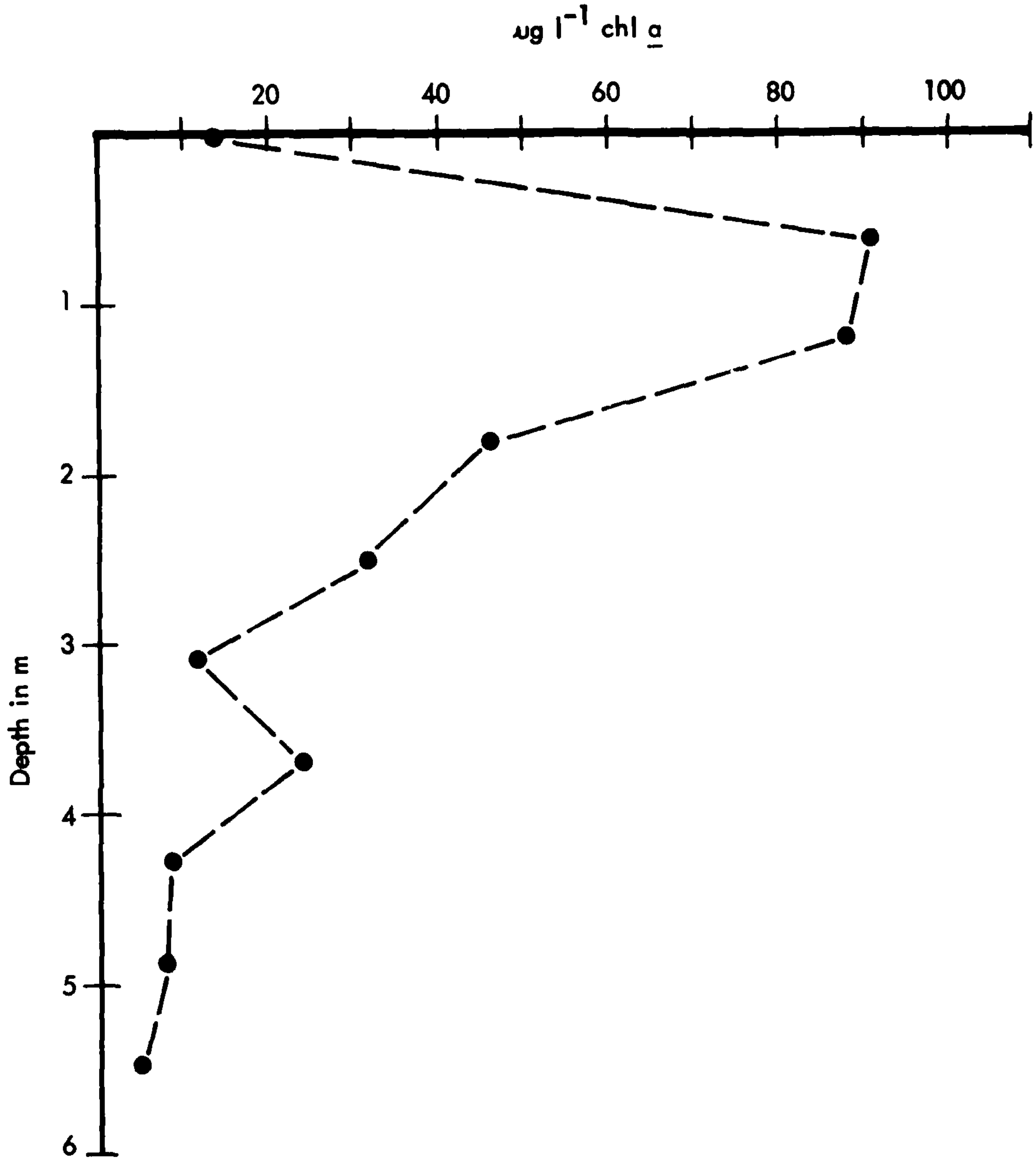


FIGURE 11

Chlorophyll a content, August 9, 1966, at 7.7 km upstream from mouth of Duwamish

TABLE 2

Biomass model predictions of chl*a* at 16th Ave. S. (km 7.7) as a function of incoming chl*a* concentrations at East Marginal Way (km 12.6, low tide) and growth rate. (Model parameters: river flow = 10 m^3/sec (350 cfs); travel time = 1.1 days; salinity at Sta. 7.7. = 10 parts per thousand; wedge chl*a* = 1μg/l).

km 12.6 chla (μg/l)	km. 7.7 chl*a* (μg/l) Growth = 2.2 day^{-1}	km. 7.7 chl*a* (μg/l) Growth = 1.5 day^{-1}
1	7.4	3.6
2 (mean summer 1965, 1966, 1968, and 1979)	14.3	6.8
3	21.3	10.1
4 (mean summer 1967)	28.3	13.3
5	35.2	16.5
6	42.2	19.8
7	49.2	23.0
8	56.2	26.2
9	63.1	29.4
10 (1966 - 67 blooms)	70.1	32.7

values nearly double those of other years on record. Inoculum and limitation factors will be addressed later in this section.

Nutritional Factors

Ascribing control of bloom timing to strictly physical and inoculum factors assumes that nutrient content is sufficiently high to be nonlimiting. Nutrient (soluble N and P) content was sufficiently high to be considered nonlimiting to growth rate even before Metro's Renton Treatment Plant began discharging effluent in 1965 and there was little evidence that nutrient content was depleted sufficiently during blooms to reach limiting levels (Welch, 1969). However, it has also been observed that water from the head of the dredged channel (vicinity of km 7.7) was more stimulatory to algal growth than water from sites located downstream or upstream (Welch, et.al.,1972). That was the case even with Metro waste effluent in the estuary during 1965-66 and algal growth appeared to be independent of N and P. The specific factor(s) reguluting algal growth was not identified. As evidence of this effect, periphytic algal accrual rates on artificial substrates, under conditions of controlled light, were on the average 17 times greater in km 7.7 water than in upstream (km 12.6) water. Apparently the water overlying the saline wedge toe contained the stimulatory factors.

Before 1970 the wedge D.O. content fell to lows of 1 mg/l. After 1970, when waste discharges to the lower estuary were diverted, oxygen did not drop below 4 mg/l (Figures 9 and 10). With the lower oxygen content in the 1960's, there may have been a release of nutrients, including trace elements, that were stimulatory. In contrast, the low D.O. content may have allowed persistence of a higher content of

dissolved organic carbon (DOC) to complex toxic heavy metals and eliminate inhibition. Further, low D.O. in the saline wedge results from a high respiration rate and therefore high CO_2. Of the three macronutrients, N, P, and C, C is most apt to deplete first and the high pH observed during blooms (9.0 in 1966) suggests severe CO_2 depletion. Thus, the higher CO_2 in the wedge water than in overlying water may account for the stimulation. Although more remote, the possibility also exists that DOC from RTP effluent aided in complexing toxic heavy metals in sea water resulting in the observed stimulation at km 7.7. This is difficult to accept, however, because periphyton accrual at km 12.6, where RTP effluent is actually more concentrated, was much less than at km 7.7 (RTP effluent more dilute), as noted previously.

Although an unexplained chemical phenomenon may account for a greater stimulation of algal growth in water overlying the saline wedge toe, it is highly unlikely that this phenomenon is linked with RTP effluent. The original assumption still holds that the content of soluble N and P in the Duwamish estuary prior to RTP effluent was sufficient to allow the formation of a bloom. This is supported by supersaturated DO in the surface water in 1963, before the RTP (Welch, 1969).

Short-term bloom limitation would have been more apt to occur from CO_2 depletion. The mid-morning raising of pH and rapidly declining CO_2 to limiting (King,1970) levels shortly after noon during the August 11, 1966 bloom coincides with an afternoon depression of photosynthesis and indicates such limitation of the bloom (Figure 12). However, chlorophyll a levels on the preceeding day were extremely large (70 μg/l) and suggests that CO_2 limitation probably only occurred during very large blooms. The

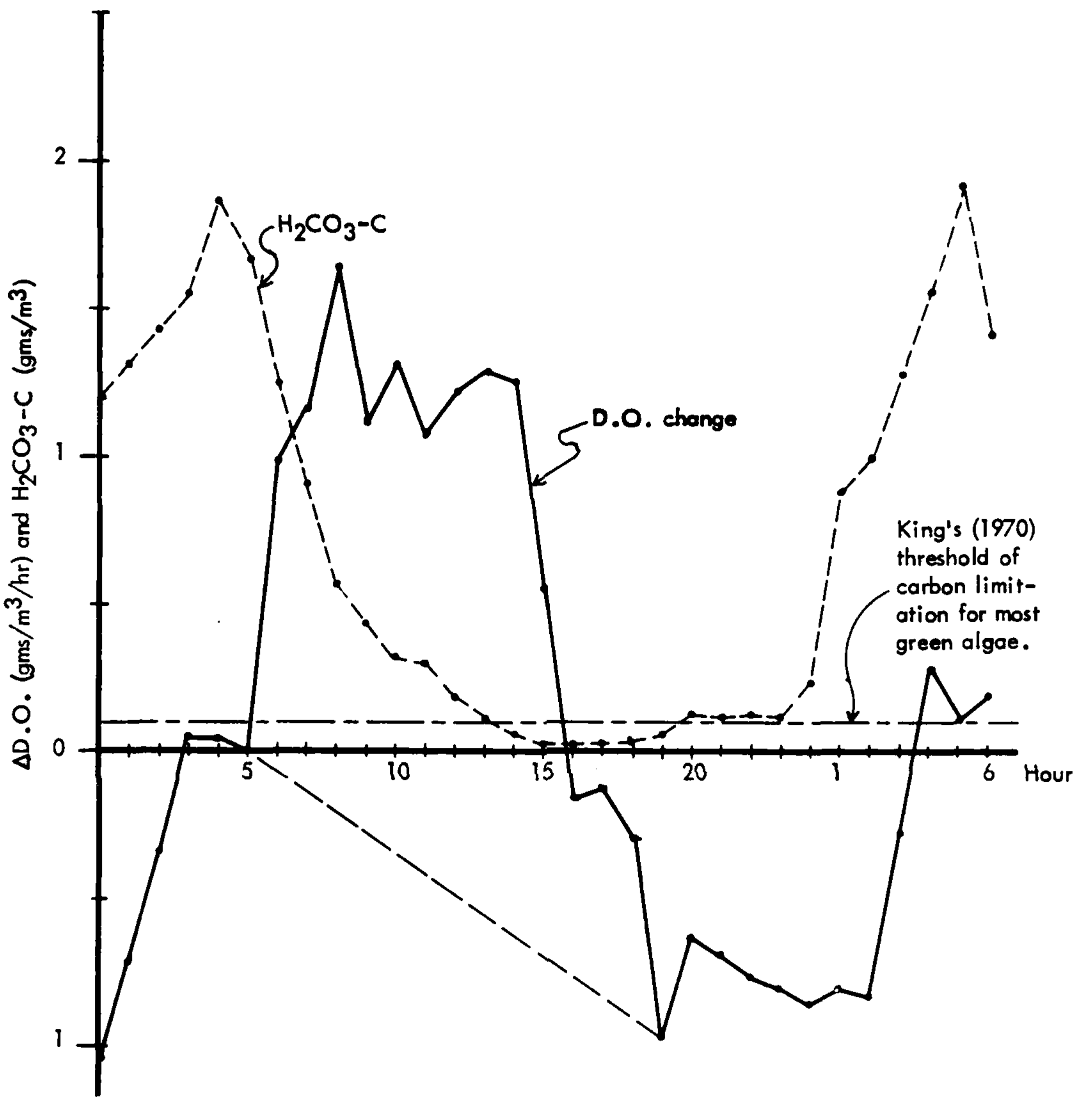


FIGURE 12

Daily variation in photosynthesis and free - CO_2 at km 7.7, surface during a bloom on August 11, 1966. Photosynthesis is gross oxygen production calculated by the free-water method of Odum (1956) using Metro monitor data and a reaeration rate = .05 hr^{-1}. Free CO_2 (H_2CO_3) was calculated using pH, temperature, alkalinity, and specific conductance measurements.

continual renewal of CO_2 laden water to the surface also minimizes the long-term importance of such limitation. If not for the rapid flushing of plankton cells out of the estuary, light availability rather than CO_2 would be the limiter to further plankton biomass increase. Although CO_2 production in the saline wedge may have decreased along with oxygen demand following sewage diversion in the late 1960's, that lost CO_2 supply does not account for the decline in blooms. The usefulness of CO_2 to continued growth would occur only after the bloom was progressing and depletion had occurred.

Trends From 1967 - 1980

The spring and summer of 1979 in the Pacific Northwest were relatively dry and very low flows persisted in the Duwamish River during most of the summer and fall. The hydrographic conditions for a phytoplankton bloom in the estuary were ideal. However, a bloom did not occur and frequent sampling from late July through September showed a chl a maximum of only 10 $\mu g\ l^{-1}$ (Welch and Trial, 1979). A review of the trend in the daily range of DO, as an index of photosynthetic activity, shows that the frequency and intensity of phytoplankton blooms at km 7.7 have declined markedly since 1965-1967 when large blooms occurred (Figure 7). During this time RTP effluent volume increased over three fold.

Possible explanations for the decline of plankton booms are: 1) toxicity of RTP effluent caused by trace metals and/or organics, 2) toxicity of RTP effluent caused by residual chlorine, 3) complexation of trace element nutrients by increasing DOC content from RTP effluent, and 4) diversion of Kent and Auburn sewage lagoon effluents with their entrained algal inoculum.

Trace metals and organic compounds in RTP effluent are probably not responsible for the decline in blooms in the estuary. Algal bioassays in vitro with effluent and river inoculum (Welch, 1969) as well as many classroom assays with unialgal cultures have shown that RTP effluent is relatively nontoxic. In many instances, in fact, results indicate that an added stimulant other than N or P may be present in the effluent. Figure 13 shows typical results. The effluent is certainly stimulatory because the biomass attained with 10% non-chlorinated effluent in Lake Washington water is higher than with the equivalent - effluent additions of inorganic N and P, indicating an added beneficial effect. Of course if effluent were collected after chlorination for use in bioassays, severe inhibition would persist for nearly a week after which growth began.

Residual chlorine may have exerted an inhibitory impact on blooms over the past 12 years, however, the evidence is not clear. Although chlorinated effluent is definitely inhibitory in vitro, the effluent has been dechlorinated since 1974 leaving a residual $TRCl_2$ of 0.25 mg/l. In fact, $TRCl_2$ loads to the Green River are presently (1979-80) one-fourth of the 1971-73 loads. Furthermore, $TRCl_2$ content is very low and undetectable by colorimetric methods in the lower estuary (km 7.7) and should not inhibit in the area of maximum productivity (Brooks and Liptak, 1978). Yake (1979) has shown that $TRCl_2$ content declined to undetectable levels in the first few km downstream from the effluent input at low river flow. Concentrations within that stretch of river, representing at most about one half day travel time, ranged from .02 to .20 mg/l and are sufficient to cause some inhibition to algae.

Complexation of trace elements with increasing DOC content from RTP effluent has been considered a possible deterrent to plankton blooms through growth limitation.

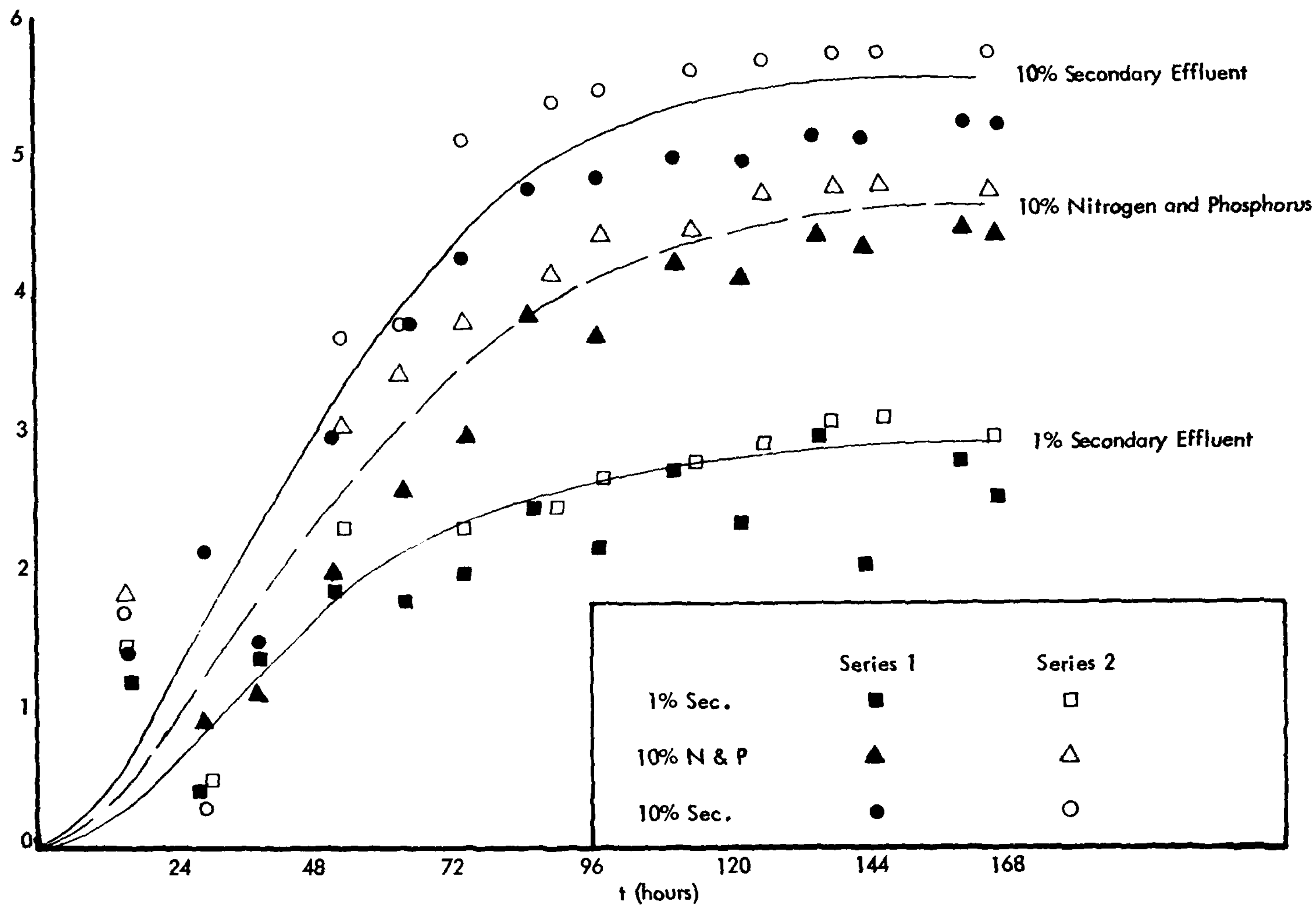


FIGURE 13 - Growth of <u>Selenastrum capricornutrum</u>, a green alga, in Lake Washington water with unchlorinated RTP *effluent* as well as effluent equivalents in inorganic N an P added (experiment performed by Robert James).

This does not seem likely in view of the fact that in vitro bioassays with a test organism (Selenastrum) adapted to high trace metal content showed no inhibition to a high concentration (10%) of effluent in water (L. Wash.) that no doubt contains less trace elements than the Duwamish. In other tests Lake Washington water can be shown to be trace element limiting, which may offer a explanation for the effluent stimulation in Fig. 13. In any event inhibition through the alternative of effluent DOC complexation of trace elements does not seem plausible.

The last possibility as a cause for diminishing estuarine blooms may be most credible. Green algae species such as, Micratinium, Scenedesmus and other unicellular greens, were abundant in the 1965 - 1967 blooms. These algae are typical of sewage lagoon flora and may have been added to the Green - Duwamish River with the discharge from the Kent and Auburn sewage lagoons. Although no cell counts or identifications were performed on the lagoon flora, chlorophyll analyses revealed a high level of chl c relative to chl a (30%). Chlorophyll c is only found in diatoms and dinoflagellates and this high percentage suggests that diatoms may have been abundant and dominant in the lagoons. The possibility exists, though unproven, that these species may have been the same diatoms (Cyclotella and/or Stephanodiscus) found in some 1966 - 67 blooms. Inoculation of the relatively stable surface water in the lower estuary with a highly viable culture of algae would have shortened the lag-time to bloom maximum (Table 2). However, growth rate in relation to light is also an important factor limiting growth. Much of the time during 1965 - 1966, the algae entering the lower estuary were of periphytic origin and did not appear viable although they were enumerated as phytoplankton if cells were in tact (Welch, 1969). Only at low flow could the lagoon inoculum have reached the estuary in significant concentrations. Mean chl a

concentrations in the lagoons were 100 - 200 μg/l and under summer low-flow conditions would theoretically raise river chl a levels (without growth) by 2 - 4 μg/l. Several measurements of river chlorophyll above and below the lagoons during the early 1970's have substantiated this increase (Harper, 1972).

Two factors may have reduced the impact of the "upstream inoculum" between 1967 and 1980. First, the one-half day or so contact of a parcel of river water containing lagoon flora with $TRCl_2$ of the effluent may have caused enough inhibition of the "upstream inoculum" to significantly slow bloom development in the lower estuary surface water. This effect would have progressively increased until dechlorination was installed in 1974. However, even after that event some inhibition no doubt still exists in the vicinity of the effluent discharge. Second, lagoon effluent was diverted from the river; the Kent lagoon in 1974 and the Auburn lagoon in 1977. A noticiable decrease in relative photosynthetic activity during those years can be seen in Fig. 7. Thus, the combination of increased $TRCl_2$ content, even if effective for only a fraction of a day, and an eventual elimination of lagoon inputs may have contributed to a significant decrease in the quantity and viability of the upstream inoculum over the period 1967 - 1980.

Polychlorinated Biphenyls

The Duwamish River estuary has been identified as the major source of PCBs to Elliott Bay (Pavlou and Dexter, 1979). Although it is not known whether the major PCB source to the Duwamish is industrial discharge, spills or dredging of previously contaminated sediments, the sediments appear to retain nearly all the present input. Water column concentrations average 22 ng/l or roughly 5 times higher than the main basin of Puget Sound. Nearly all of the water column concentration is due to PCBs associated with suspended particulate matter, and levels in the overlying mixed water are considerably greater than levels in

the salt wedge. PCB concentrations in Duwamish sediments are large with respect to national levels and roughly 18 times more concentrated than the main basin of Puget Sound. A portion of this load would probably be released and/or redistributed in the Duwamish River following sediment agitation (e.g. dredging).

Sediment Quality

Concentration of most parameters in the sediments are generally highest in the region from the Head of Navigation to 16th Ave. S., where most sediment deposition occurs. Analysis of sediment data collected over a twelve-year span revealed no significant trends in any of the parameters. Table 3 presents average concentrations of selected parameters from cores taken in regions of "deposition" and "stability" (relative) from 1969 to 1980.

TABLE 3

Historical average sediment concentrations of selected parameters from samples collected from the Boeing Bridge to 16th Ave. So. (deposition zone) and from 1st Ave. So. to Spokane St. (stable zone).

	deposition zone	stable zone
Volatile solids (%)	8.2	5.5
COD (gm/kg)	76.	57.
Pb (mg/kg)	18.	23.

The higher organic content of sediments collected from the "deposition" region agrees well with longitudinal variations in sediment oxygen demand (SOD) rates taken throughout the estuary in August, 1973 (Figure 4) (41 determinations) (R.I. Matsuda, written communication, 1975). This information suggests that major organic inputs to the Duwamish

arise from upstream sources (e.g. RTP), although a much more comprehensive study would be required to test this result.

Other Factors Influencing Water Quality

Given the importance of adequate dissolved oxygen conditions in the Duwamish and the substantial quantity of information available concerning oxygen in the estuary, this section will principally focus on factors influencing D.O. regimes in the Duwamish. In addition, Haushild and Stoner (1973), with the use of the USGS model, have stated that D.O. conditions in the salt wedge would deteriorate as a result of the "large channel" dredging project (refer to chapter entitled "Impact of Dredging Project" for a description of the channel alternatives). Insomuch as many of the more common wastes require oxidation, the variation in D.O. concentrations is a general indicator of the persistence of these types of pollutants. Previous sections have provided an overview of various problems in the estuary, and in subsequent sections these problems will be discussed with respect to the proposed dredging project.

Circulation/Flushing

When pollutants are discharged into the estuary, one of the most important factors determining impacts is flushing rate. Rapidly flushed systems are generally able to tolerate waste discharges better than more quiescent systems. Much of the hydrodynamic work performed by the USGS ultimately led to a description and an attempt at quantification of the processes which affect flushing rate. Obviously, one of the more important factors is freshwater discharge, and indeed the work of Stoner (1972) verified that flushing rate in both the surface and mixed layers increased with river discharge. The other driving factor which influences flushing rate is tidal exchange or tidal prism, and this factor dominates entrainment

flows at river discharges less than 25 m^3/sec (880 cfs) (Prych et.al., 1976). A larger tidal prism thickness enhances entrainment discharge from the wedge to the surface layers. This phenomenon was also observed by Stoner (1972). The effect of tidal prism thickness on wedge travel time can be illustrated with output from the USGS numerical model (Fig. 17). Variations in tidal prism during a nearly constant low-flow period can result in nearly two-fold variations in residence time.

The results of both the USGS model and a mass-balance model (described above) indicate that throughout the summer of 1967 rates of oxygen consumption in the wedge varied little from week to week. Yet an examination of mean (or minimum) D.O. concentrations at km 7.7, bottom reveals a pronounced periodic variation very much correlated with mean daily tidal prism thickness (Fig. 9). Freshwater discharge remained relatively constant and low. Periods of minimum D.O. coincide with low tidal exchange periods, with corresponding increased residence time in the wedge. Given a constant oxygen consumption rate, the increased residence time (less wedge discharge) would therefore result in a lower D.O. concentration at the wedge toe. The same phenomenon is not as evident in 1976 simply because the oxygen consumption rate is roughly 25 percent of the 1967 rates, thereby smoothing the curves considerably (Figure 10). A similar situation of tidal prism effects on flushing time also exists for the surface layer, and can be demonstrated both numerically (USGS model) and with phytoplankton data (see previous section).

Incoming Concentrations

As mentioned previously, the concentration of oxygen at km 7.7, surface is presently being affected by a reduction in O_2-content of the inflowing freshwater. Likewise, D.O.

concentrations throughout the salt wedge portion of the estuary are influenced to a large extent by the quality of incoming seawater. Daily maximum concentrations at km 1.9 (Spokane Street), bottom (editing out occasional freshwater values) provide an approximation of the D.O. content of the seawater inflow, since maxima typically occur at high tides, when the water has had very little contact with respiratory sediments and/or suspended-dissolved materials. Examination of Figures 9 and 10 reveals that the "inflowing" water has a substantially higher oxygen content during June and July than August and September. This lower seawater "inflow" concentration in late summer appears to be the principal reason why wedge D.O. levels also reach a minimum during the same period. Apparently wedge oxygen consumption remained relatively constant throughout the summers of 1967 and 1976. Phytoplankton blooms were also more prevalent during the low flow months of August and September in past years and may significantly elevate SOD rates, depressing D.O. concentrations in the wedge (See Figure 8). Both "inflow" concentration and bloom conditions have historically resulted in minimum D.O. concentrations during the August - October period.

Comparison of daily maximum D.O. at km 1.9 bottom with observed concentrations at the 5 m depth in Elliott Bay reveals that the differences are not significant ($P=.05$). In addition, comparison of August - September "inflow" measurements between years (1967 - 1976) also yields non-significance ($P=.10$). For example, the mean August - September "inflow" concentration in 1976 was only 0.15 mg/l higher than 1967. It therefore appears reasonable to conclude that the improvement in D.O. conditions following 1969 sewer hookups was principally due to changes in O_2 consumption within the estuary.

SEDIMENT TRANSPORT

Estimates of sediment loads to the Duwamish estuary are necessary to determine the location and magnitude of shoaling. Maintenance dredging requirements may be estimated based on this knowledge of shoaling. Annual sediment loads are estimated in the following paragraphs and compared with historical maintenance dredging of the waterways.

Sources of Sediment

Sources of sediment to the Duwamish estuary include natural and cultural sources upstream from the Duwamish River, the Renton treatment plant, combined sewer overflows and local stormwater runoff generated in the watershed downstream from Renton Junction. Records are available to estimate historical loadings from watershed sources upstream from Renton Junction, the RTP and CSO's. Many assumptions are required to estimate sediment loads from stormwater runoff which is considered a minor source based on loadings presented in the following sections.

Estimation of Annual Sediment Loads

Suspended sediment transported to the Duwamish estuary from sources upstream from Renton Junction appears to be related to the magnitude of freshwater inflows. The U.S. Geological Survey measured suspended sediment and freshwater inflows at Tukwila for nearly three years (October 1963 through June 1966). Suspended sediment loads near 30,000 tons per day were measured during flows exceeding 300 m^3/sec. (10,000 cfs). Lowest suspended sediment loads occur during low freshwater inflows.

Analysis of the USGS data indicates that daily suspended sediment load appears to be related to freshwater inflow. Daily suspended sediment loads were correlated with average daily discharge at Tukwila using data reported by the USGS for suspended sediment loads in excess of 500 tons per day (176 observations). An exponential relationship was developed having a correlation coefficient of 0.83.

The average annual sediment load from upstream sources may be estimated using the relationship in Figure 14 knowing the average annual frequency of occurrence of streamflows at Tukwila. Frequency analysis of daily streamflows at Tukwila was undertaken for the period of record since regulation of flows in 1961 by Howard A. Hanson reservoir. The results of this analysis is shown in Figure 15. Frequency distributions of streamflows at the gaging station near Auburn are also shown for unregulated and regulated flow conditions. The distributions for both periods at the Auburn gage are similar which suggests that the frequency distribution for the Tukwila flows should be similar to unregulated flows.

Average annual sediment loads from upstream sources are computed in Table 4 using suspended sediment-flow relationship and the frequency distribution of flows at Tukwila. The annual suspended load is estimated to range from 190,000 to 226,000 tons per year, depending upon the value of flow used to represent the flow class in the computations. Using a density of 90 pounds per cubic foot, the average annual suspended sediment load ranges from 120,000 m^3 (157,000 yd^3) to 142,000 m^3 (186,000 yd^3).

Suspended sediment loads from all other sources are minor relative to the estimated upstream load. The Renton Treatment plant is estimated to presently discharge approximately 800 m^3/year (1050 yd^3/year) which would increase to 2000 m^3/year (2600 yd^3/year) for wastewater flows expected in year 2000. Suspended sediment loads from existing combined sewer overflows are estimated at 250 m^3/year (330 yd^3/year) and should be less in the future.

Bedload in the Duwamish estuary has been estimated by the Corps of Engineers to range from 38,000 m^3/year (50,000 yd^3/year) to 61,000 m^3/year (80,000 yd^3/year). Total

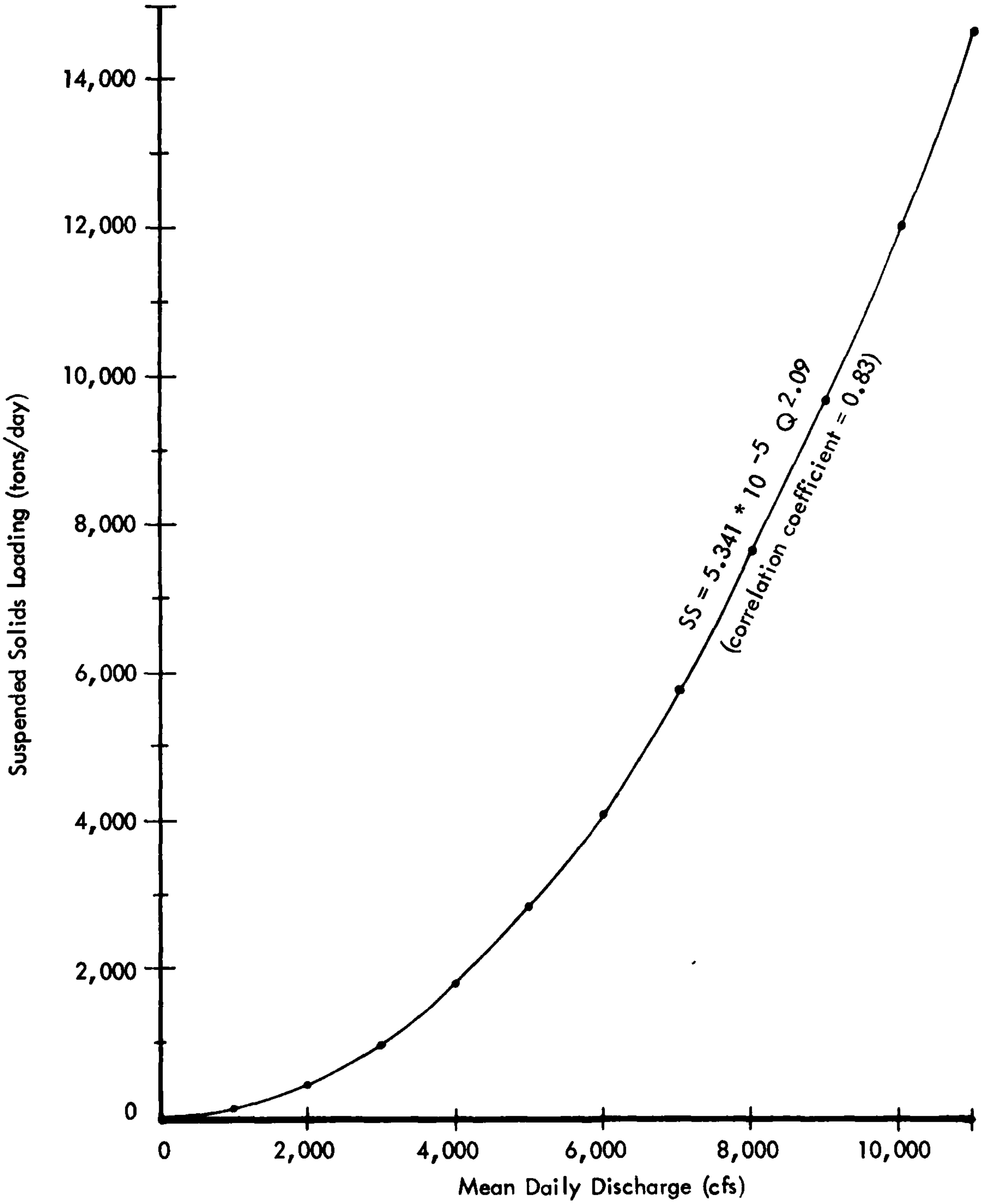


FIGURE 14

Suspended Sediment Load vs. Discharge at Tukwila

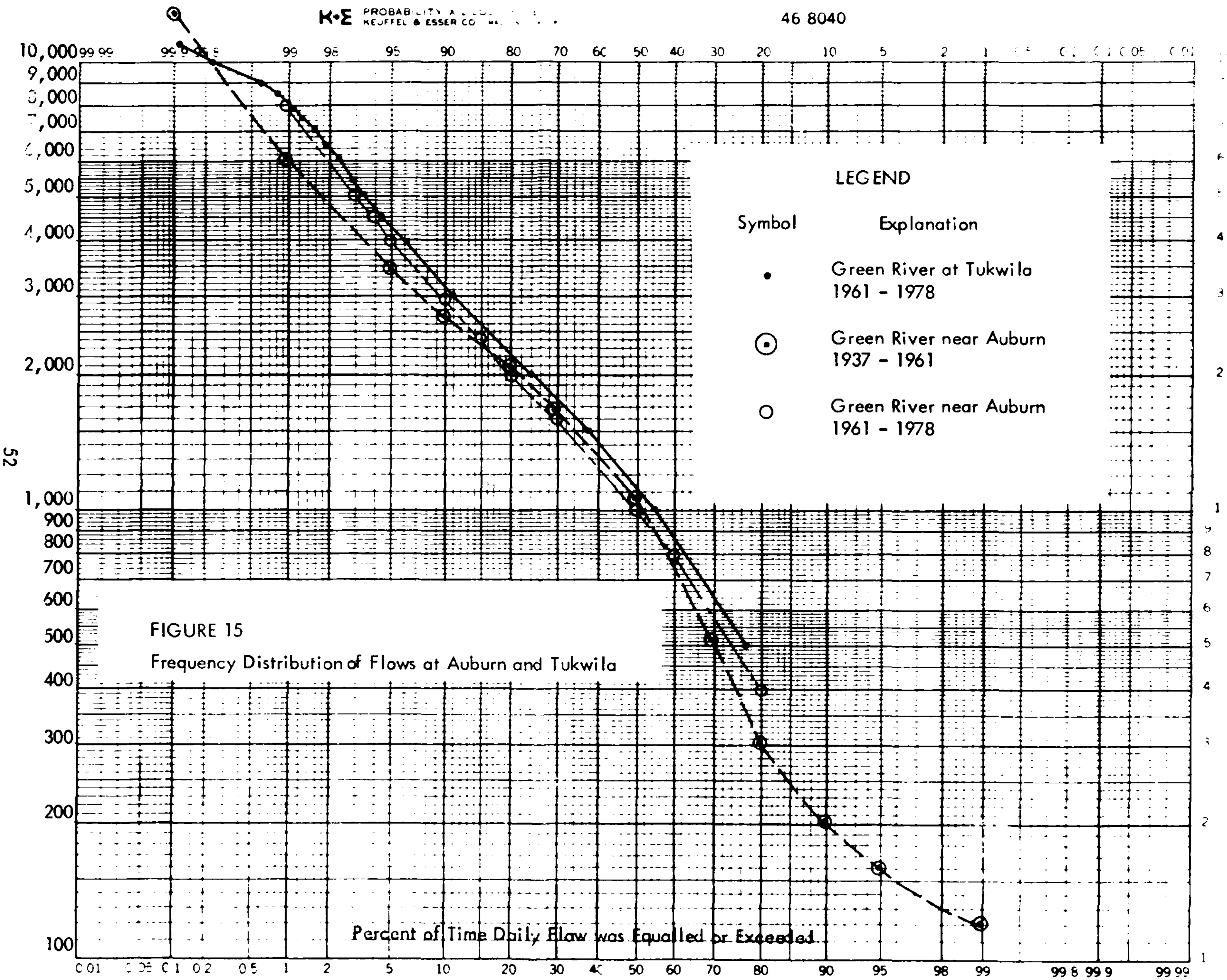


FIGURE 15

Frequency Distribution of Flows at Auburn and Tukwila

TABLE 4

Computation of Annual Suspended Sediment Load in Green River

Flow Class	Mean Q (cfs)	S.S. Load (tons/day)	Ave. Days Per Year	Class S.S. Load (tons/yr)	Load Using Upper Limit of Class
0- 500	250	5.5	83.95	461	1,961
501- 1,000	750	54.5	82.86	4,517	8,241
1,001- 1,500	1,250	158.6	62.05	9,838	14,401
1,501- 2,000	1,750	320.3	47.82	15,317	20,248
2,001- 2,500	2,250	541.6	30.30	16,411	20,453
2,501- 3,000	2,750	823.8	18.62	15,339	18,399
3,001- 3,500	3,250	1,168.0	10.95	12,790	14,933
3,501- 4,000	3,750	1,575.	6.21	9,782	11,195
4,001- 4,500	4,250	2,046.	6.21	12,707	14,320
4,501- 5,000	4,750	2,582.	3.65	9,423	10,490
5,001- 5,500	5,250	3,182.	2.56	8,147	8,979
5,501- 6,000	5,750	3,849.	2.19	8,429	9,213
6,001- 6,500	6,250	4,582.	1.46	6,689	7,260
6,501- 7,000	6,750	5,381.	1.46	7,856	8,477
7,001- 7,500	7,250	6,249.	0.73	4,561	4,895
7,501- 8,000	7,750	7,182.	0.73	5,243	5,603
8,001- 8,500	8,250	8,185.	1.10	9,003	9,583
8,501- 9,000	8,750	9,256.	0.73	6,757	7,167
9,001-10,000	9,500	10,992.	1.17	12,860	14,316
10,001-11,000	10,500	13,549.	0.51	6,910	7,615
11,001-12,000	11,500	16,386.	0.44	7,210	7,880
			365.7	190,250	225,630

average annual sediment load may range from 158,000 m^3/yr (210,000 yd^3/yr) to 203,000 m^3/yr (266,000 yd^3/yr).

Comparison of Annual Sediment Loads with Dredging

Corps of Engineers records were reviewed to obtain maintenance dredging quantities for the period 1960 to 1980. Dredging quantities for Port of Seattle projects also are available for the period 1965 to 1979. The Port of Seattle projects include excavation of material for land reclamation and new berthing areas in addition to maintenance dredging. Data from both sources is summarized in Table 5.

Annual sediment loads should be compared with annual maintenance dredging quantities. Based on review of the Port of Seattle records, it is assumed that 21,200 cubic yards in the lower Duwamish Waterway and approximately forty percent of the dredged material near the head of navigation was removed for maintenance of the channel. These quantities plus the Corps of Engineers quantities amount to at least 2.8 million m^3 (3.7 million yd^3) dredged in 20 years for maintenance of the waterways (Table 6).

The total maintenance dredging of 2.8 million m^3 is equivalent to an annual rate of 143,000 m^3/yr (186,900 yd^3/yr). The total annual sediment loading to the estuary was estimated to range from 158,000 to 203,000 m^3/yr. Using these estimates, the proportion of input which settles in the estuary ranges from approximately 70 to 90 percent of the incoming sediment load. Settling of eighty percent of incoming sediments would seem reasonable based on settling characteristics of the sediments and favorable conditions for settling provided by the saline estuary.

Shoaling Rates

The majority of the sediments presently settle in the reach between 16th Avenue South and the head of navigation (Table 6). Sediment transported in freshwater encounters

TABLE 5

Summary of Quantities Dredged from Duwamish Estuary [1] (cubic yards)

	Port of Seattle Projects [2]	Corps of Engineer Projects	Total
East Waterway	668,300	-	668,300
West Waterway	46,100	-	46,100
Duwamish Waterway			
0+00 to 1st Avenue S.	1,228,100	303,000	1,531,000
1st Ave. S. to 16th Ave. S.	-	304,700	304,700
16th Ave. S. to Head of Navigation	325,100	2,941,500	3,266,600
	2,267,600	3,585,200	5,852,800

(1) Port of Seattle project 1965 - 1979.
Corps of Engineers project 1960 - 1980.

(2) Volumes estimated based on density of 90 pounds/ft^3.

TABLE 6

Estimated Maintenance Dredging in Duwamish Waterway
(Quantities in cubic yards; 1 cubic meter = 1.308 cubic yards)

	Port of Seattle	Corps of Engineers	Reach Total	% of Waterway Total	Annual Equivalent
0+00 to 1st Ave.S.	21,700	303,000	324,700	8.7	16,300
1st Ave.S. to 16th Ave.S.	-	340,700	340,700	9.1	17,000
16th Ave.S. to Head of Navigation	131,700	2,941,500	3,073,200	82.2	153,600
	153,400	3,585,200	3,738,600		186,900

saline water in this reach which promotes sedimentation. Moreover, the estuary becomes wider and deeper relative to the undredged portions which results in slower velocities more conducive for settling of sediments.

IMPACT OF DREDGING PROJECT

DESCRIPTION OF DREDGING PROJECT

Three alternatives have been considered by the Corps of Engineers for construction (Table 7). These alternatives range from a small channel project which is presently authorized to a project involving removal of 4.7 million cubic yards. The mid-sized channel project is most likely to be recommended for construction according to discussions with Corps personnel. Consequently, evaluations of impacts have been undertaken for the mid-sized channel alternative. The evaluations have included three-feet additional depth for advance maintenance and contractor overdepth allowance. Typical cross-sections for the existing channel and dredging alternatives are shown in Figure 16.

TABLE 7

Description of Channel Alternatives (ft.)

Reach	Small Channel	Mid-Sized Channel	Large Channel
East Waterway	750 x 34	550 x 40	550 x 45
West Waterway	750 x 34	550 x 40	550 x 45
Duwamish Waterway to 1st Avenue S.	200 x 30	250 x 35	250 x 40
1st Ave. S. to 8th Ave. S.	150 x 20	200 x 20	250 x 20
8th Ave. S. to Head of Navigation	150 x 15	200 x 20	250 x 20

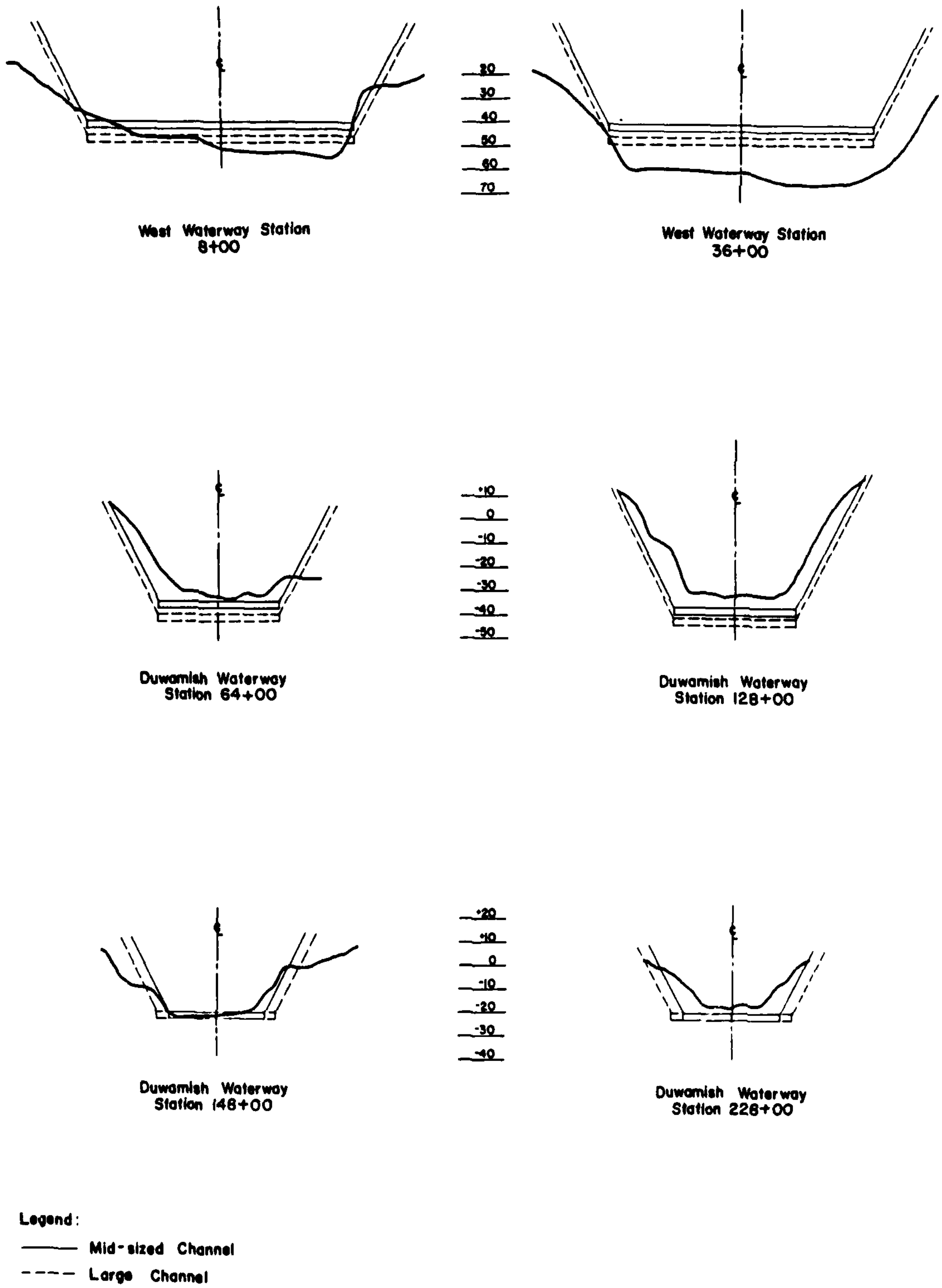


FIGURE 16

Typical cross-sections of the existing and proposed (mid-channel) Duwamish estuary. Cross-sections have three-feet additional depth for advance maintenance and contractor overdepth allowance.

EFFECTS OF PROJECT ON ESTUARY HYDROLOGY

The medium-sized channel project will result in increased channel sections relative to the existing waterway sections. The increased sections may provide for increased entrainment from the wedge to the upper layer, but the increase in flow in the wedge may not compensate for increased volume. Consequently, detention times in wedge may increase with subsequent impacts on water quality. These hydrologic factors are evaluated in the following sections for low and high river flows and future conditions in the Green River watershed using the mathematical model developed for the Duwamish estuary by the USGS.

General Description of the Model

The Duwamish estuary model accounts for water movement and constituent transport in both the surface layer and saltwater wedge. Input to the model includes estuary geometry, tidal data, freshwater flow, climatic and weather conditions, water quality parameters and corresponding growth/decay rates, and diffusion and dispersion coefficients. The main thrust of the model is in predicting water quality changes based on estuarine flow conditions. However, in this application, the model has been used only to assess physical differences in circulation and saltwater residence time based on a changed estuary geometry for the proposed mid-sized channel and various freshwater flows.

The model uses tidal data, weather conditions, freshwater inflow, and empirical hydraulic relationships to calculate wedge location, volume, and residence time. The mouth of the estuary is defined at 1067m (3500 ft) downstream from the Spokane St. bridge and the toe of the wedge moves with tidal fluctuations. Since the model assumes that the only route of water loss from the wedge is through entrainment into the upper layer, the

wedge residence time is directly related to entrainment velocity. By empirical relationship, entrainment velocity increases with tidal prism thickness (vigorous tidal actions) and freshwater flow, and is directly proportional to the top surface layer of the wedge. Therefore, either vigorous tidal actions or high freshwater flow will produce higher entrainment velocities which in turn contribute to shorter wedge residence times. Conversely, slow tidal action or low freshwater flow produce longer wedge residence times. Another critical factor is the wedge geometry. As explained earlier, an enlarged estuary, particularly a deepened estuary, may further lengthen wedge residence times by increasing the wedge volume without a proportionate increase in entrained flow.

Conditions for Assessment of Impacts

Baseline weather and tidal dates used in the simulations were developed from actual conditions that occurred between 16 August 1967 and 15 September 1967. This late summer period is typically a critical time for water quality in the Duwamish estuary, and a large algal bloom did occur at that time in 1967. This critical time period was selected to confirm that the physical flows in the estuary predicted by the model were consistent with those known to be favorable to algal blooms, and to develop a conservative, worst-case baseline with which to compare the proposed project's impact.

Reference cross sections for the existing estuary were taken directly from Corps of Engineers' condition surveys determined in 1978. Reference cross sections for the proposed, mid-sized channel were derived from data supplied by the Corps.

Several low flow conditions may occur in the future depending upon implementation of several projects such as diversion of the Renton treatment plant discharge to the Duwamish River. Existing low flows immediately downstream from the discharge of RTP

effluent is approximately 9.3 m^3/sec (330 cfs): low flow in Green River is 7.8 m^3/sec and existing RTP flow is 1.5 m^3/sec.

The Washington State Department of Ecology recently established an instream flow of 8.5 m^3/sec (300 cfs) at Auburn during summer months of normal water years (Chapter 173-509 WAC). Tributary drainage between the Auburn gaging station and Tukwila appears to contribute approximately 1.4 m^3/sec (48 cfs) during low flow months. New low river flows at Tukwila upstream from the RTP discharge will be 9.9 m^3/sec in the future. If the RTP discharge to the Duwamish River continues until 1987 (estimated completion date of outfall line to Puget Sound), flows from the RTP will have increased to 2.3 m^3/sec (82 cfs). Consequently, low freshwater inflows to the Duwamish estuary may be 12.2 m^3/sec. For the purposes of the following hydrologic evaluations, the range of low flows will be 9.3 to 12.2 m^3/sec.

Location of the toe and residence time of the wedge were also evaluated for a high flow condition. A flow of 84.5 m^3/sec (3000 cfs) was used.

Results of Simulations

Table 8 summarizes the results of month-long simulations of both the existing and proposed mid-sized channel geometries at freshwater flows of 9.3, 12.2 and 84.5 m^3/sec (330, 430, and 3000 cfs). Wedge toe location, relative to the Spokane Street bridge, is affected only by changes in freshwater flow. The results show that by increasing the freshwater flow from 9.3 to 12.2 m^3/sec, the wedge is displaced an average of 460 m (1500 ft) down the estuary. However, the maximum wedge toe position under high tidal conditions is only changed 190 m (600 ft) downstream. At much higher freshwater flows, the wedge toe is pushed significantly further down the estuary. A flow of 84.5 m^3/sec

TABLE 8

Summary of Duwamish Estuary Simulation for Critical 1 Month Period

Tidal Prism Thickness, m		
	maximum	6.4
	minimum	2.8
	average	4.6

	9.3 m^3/sec (330 cfs)		12.2 m^3/sec (430 cfs)		84.5 m^3/sec (3000 cfs)	
Wedge Toe Location,[1] m (ft)						
maximum	8720	(28600)	8530	(28000)	7060	(23200)
minimum	4850	(15900)	4850	(15900)	4840	(15870)
average	7800	(25600)	7340	(24100)	6130	(20100)
	existing[2]	dredged[4]	existing[2]	dredged[4]	existing[3]	dredged[4]
Residence Time, Days						
maximum	10.7	12.0	9.9	11.6	5.9	4.7
minimum	4.9	5.6	4.8	5.2	2.7	2.7
average	7.6	8.7	7.2	8.3	3.2	3.7

1 above Spokane St. Bridge

2 8/16 - 9/10

3 8/16 - 9/1

4 Proposed Mid-sized channel configuration

(3000 cfs), for example, is projected to displace the toe an average of 1670 m (5500 ft) down the estuary or 1660 m (5400 ft) under high tidal conditions. The difference in maximum and average wedge toe displacement is small under high flow, presumably due to the steeper slope in the estuary at that location.

The proposed project's impact on wedge residence time at different flows is also shown in Table 8 . Residence time here is defined as the length of time it takes a particle of water entering the estuary mouth to reach the toe, assuming that it has not been entrained into the upper layer. Under low flow and worst-case tidal conditions, the proposed project is projected to increase the average residence time of water in the wedge from 7.6 to 8.7 days, an average of 1.1 days. (Fig. 17). The maximum residence time in the dredged estuary at low flow is expected to be 12days.

Increases in freshwater flows minimize the impact of dredging on residence time. The model predicts that flow enhancement from 9.3 to 12.2 m^3/sec will change the average residence time to 7.2 and 8.3 days for the existing and dredged conditions, respectively. A very high flow, however, will cause a significant decrease in residence time in either the existing or dredged estuary but the difference between the two conditions will be insignificant (an average change from 3.2 to 3.7 days at 84.5 m^3/sec).

<u>Summary</u>

Simulations of estuary hydrology under critical weather, tidal, and flow conditions predict that the proposed mid-sized channel dredging project will lengthen wedge residence time an average of 1.1 days, from 7.6 to 8.7 days which is an increase of approximately 14 percent. Maximum wedge residence time under these low flow conditions (9.3 m^3/sec) is projected at 12 days. Flow enhancement to 12.2 m^3/sec will not significantly affect wedge residence time or location. Very high flows, however, will decrease wedge residence time and wedge size.

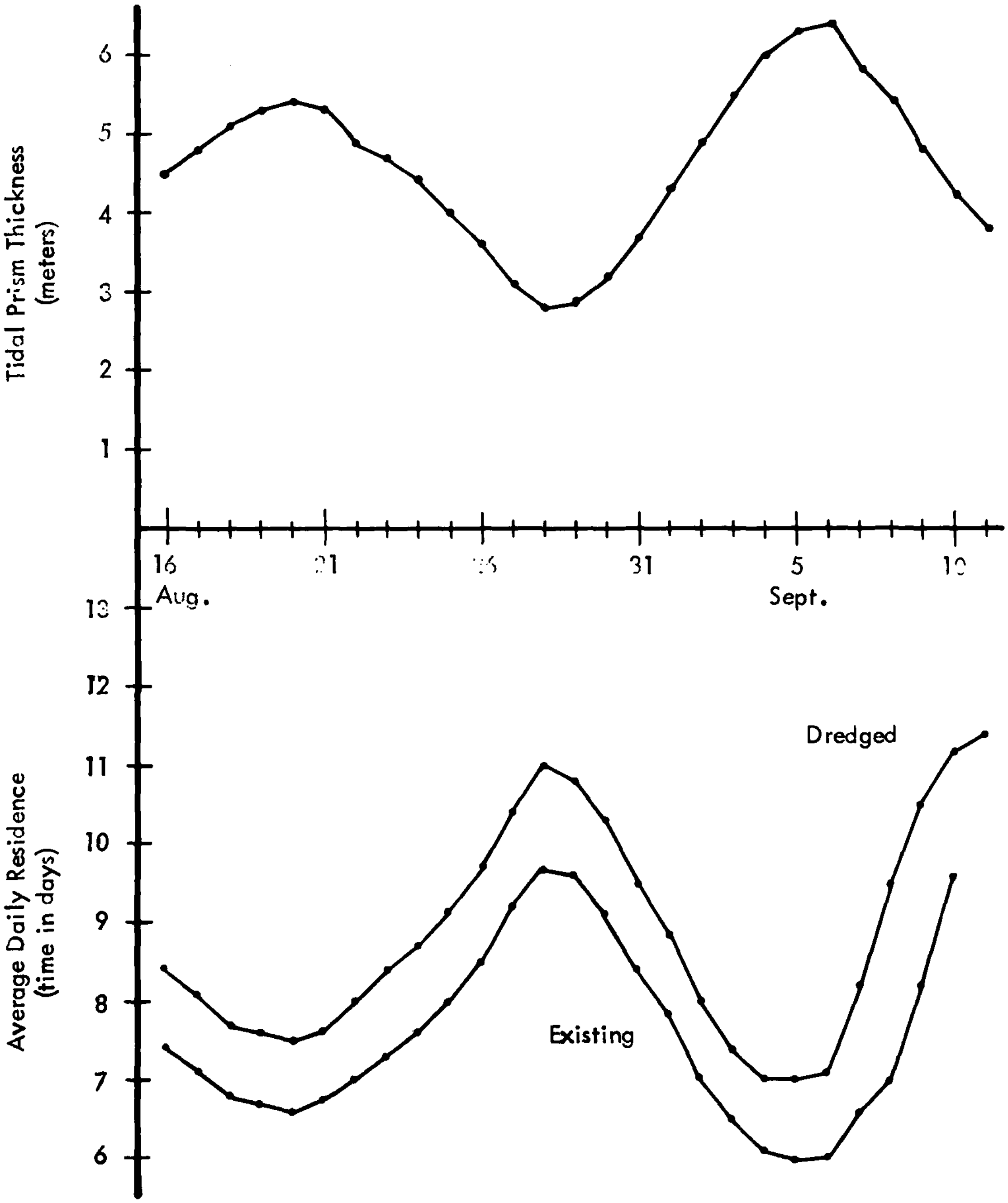


FIGURE 17

Critical Period Simulation - Saltwater

Wedge Residence Time

WATER QUALITY IMPACTS

Short Term Impacts

Short term impacts of maintenance dredging activities in the Duwamish have been investigated in several Army Corps of Engineers (ACE) studies (Baumgartner et.al. 1978). Elevations in concentrations of certain parameters (manganese, ammonia, suspended solids) have been observed during dredging but the maximal levels recorded were below ACE recommended levels and generally persisted for very short durations (minutes). Monitoring of PCB concentrations during dredging events have failed to detect increases in water column levels, but laboratory tests suggest that significant releases may in fact occur (Pavlov and Dexter, 1979). Short term impacts of dredging could be mitigated by performing the operation during periods of low biological activity (e.g. algal-induced pH increases exacerbating high ammonia levels). Periods should be selected which minimize exposure of valuable fish species (esp. salmonids) to (perhaps) elevated toxin concentrations in the dredged locale.

Impacts On Phytoplankton

Widening and deepening the dredged channel is not expected to increase the volume of the top 2m. This layer is important to phytoplankton production in the dredged portion of the estuary. However, the wedge would increase in volume by 14%. Detention time of the 2m surface layer should not change, but that of the wedge should increase by the same 14%. Entrainment, or rate of incorporation of saline wedge water into the surface layer, is not expected to change. If it would increase then the added salt would probably provide a higher gradient in the surface 4m causing increased stability for phytoplankton. This is not expected to happen, so there should be little overall difference in the light conditions for plankton algae by increasing the depth and width of the saline wedge only.

Assuming that stability of the surface layer does not increase with widening and deepening, then the potential, hydrographically, for plankton blooms will remain similar to past conditions. If the RTP effluent is diverted and the chlorine residual is removed, the blooms may reappear in late summer whether the lower estuary is enlarged or not. With that suggestion goes an assumption that the decline in blooms has been caused more by inhibition of the "upstream inoculum "than by the quantity of that inoculum. That is, blooms would still occur if it were not for the inhibition to that inoculum from $TRCl_2$ as river water passes the RTP outfall. This is clearly a hypothesis without much support. Needed to test the hypothesis are <u>in vitro</u> bioassays with "dechlorinated" effluent - river water (Renton Junction upstream of discharge) mixtures using test unialgal cultures as well as with unchlorinated effluent.

For the same reasons that the estuary widening and deepening is not expected to affect the magnitude and frequency of phytoplankton blooms, it is also not expected to alter effects of RTP - originated ammonia on fish. Ammonium content of the river and estuary has increased markedly since RTP began discharging effluent in 1965. Much of this increase has come since 1973 (Fig. 2). Concentrations as high as 5 mg/l occur in the upper river (km 16) near the discharge and average over 2 mg/l during low flow (Welch and Trial, 1979). The toxicity of ammonium (NH_4^+) is enhanced markedly by pH, roughly <u>doubling</u> with each pH unit because the fraction of unionized ammonia also <u>doubles</u> . Until the mid 1970's phytoplankton blooms during low flow caused pH to rise as high as 9.0. That coupled with high NH_4^+ due to poor dilution, would result in high unionized NH_3 fractions and marked increased in toxicity. This is at a time

(August - September) when returning salmon begin to move through the estuary and river and could cause a fish kill. However, pH during this period has decreased slightly, probably because photosynthetic activity has also declined. The trend of declining photosynthetic activity has decreased the probability of salmon migrating through pH 9 water with high NH_3 from the RTP effluent. Because the cause for bloom decline is not known there can be no assurance that blooms and high pH will not return, so caution is urged with continued NH_4^+ discharge. The potential for NH_3 toxicity will continue to exist as long as the RTP continues to discharge effluent with high NH_3 concentrations.

Impacts on Dissolved Oxygen

In an effort to determine impacts resulting from a proposed dredging operation similar to the "large channel" project under present consideration, Haushild and Stoner (1973) applied the USGS numerical model of Duwamish Waterway circulation and water quality to the estuary with modified channel configurations. The results of the model run indicated that residence time of wedge water under critical low river flow conditions would increase from 6.3 days to 8.6 days as a result of the project (37% increase). Assuming that oxygen consumption rates in the proposed wedge equal those of the existing wedge (no benthic respiration), the summer wedge D.O concentrations at 16th Ave. So. would be depressed an average of 0.4 mg/l below existing (1970) conditions with that project.

Given that the bulk of wedge respiration appears to be associated with the sediments, however, a change in residence time (with constant entrainment) will not result in any change in wedge D.O. This is due to the assumption that the increased volume is caused by increased depth. In reality, though, the surface area of the wedge will increase slightly as a result of the dredging project presently being considered. For the mid-sized channel

alternative under summer low flow conditions (9.3 m^3/sec), the surface area of the wedge will increase by 1.6%, and the mean wedge depth will increase by 12%. However, since the bulk of the increase in volume will occur as a depth increase, the relations presented above should hold true in the Duwamish Waterway if wedge oxygen consumption occurs principally as a benthic process.

August - September SOD rates from 1970 to 1976 calculated with the mass-balance model average 1.6 gm/m^2/d (SD = 0.5 gm/m^2/d for 14 monthly averages). If 10% of this rate is assumed to be associated with suspended-dissolved BOD, then average August-September BOD_w = 0.022 gm/m^3/d. A change in residence time of 37% calculated by Haushild and Stoner (1973) for a "large channel" configuration would correspond to a change in D.O. at the wedge toe of only 0.05 mg/l. Changes resulting from the mid-sized channel evaluated here (14% increase in detention time) would correspond to changes in wedge toe D.O. of 0.02 mg/l. The 1.6% increase in wedge area for the mid-sized project, assuming constant SOD rates, would correspond to a SOD induced reduction in D.O. of 0.003 mg/l.

The above calculations refer to average August - September conditions within the wedge. However, from the standpoint of periodic D.O. depletion impacts of the project, the most critical condition occurs when wedge water experiences temporary elevations in water-column BOD during periods of maximum residence time. During these periods wedge toe D.O. may be very sensitive to relatively minor changes in residence time. The blooms of the late 1960's - early 1970's observed in the Duwamish represent the largest periodic BOD source capable of depleting wedge D.O., especially since these blooms generally occur during periods of minimal tidal exchange (long residence time). During 1966 blooms,

average wedge BOD_5 values as high as 6 mg/l (200% increase) were reported (Welch, 1969). The source of this elevated BOD was sinking phytoplankton cells. Since the project is not expected to have any significant impact on phytoplankton production within the estuary, the algal-related BOD load will also remain unchanged. An increase in wedge depth will increase the length of time that sinking algal cells are suspended within a quiescent wedge, but in a well-mixed wedge (more likely) residence time would not change. But the increased wedge volume will dilute the algal cells resulting in a lower BOD which would balance out the longer residence time. The net result would be no change in DO. If the estuary is in fact quiescent with respect to algal sinking rates, then the project would tend to result in a lower D.O. concentration at the wedge toe. However, little information is available on dispersion characteristics of the wedge and settling velocities of Duwamish phytoplankton to test these hypotheses.

The question of phytoplankton blooms even occurring following removal of RTP discharge to the Duwamish (e.g. Cl_2 toxicity to phytoplankton) is of course debatable, since it appears that control of upstream sources of algal inoculum may in fact be responsible for observed declines of algal blooms. In this case, bloom effects on wedge D.O. and its importance relative to the dredging project are negligible. However, even in the event that the blooms returned, the impact of an enlarged wedge on minimum D.O. levels is probably minor, since an effect would only occur if the wedge were relatively non-turbulent. If sinking rates were the dominant factor, then transfer of organic material to the sediments would be rapid and respiration would therefore be principally benthic, thereby negating the impact of the dredging project. It is likely that D.O. depressions occuring after an algal bloom (or CSO event) would only be slightly more extreme (0.1 mg/l at most)

following dredging, but a quantitative analysis of just how severe this depression would be is hampered by lack of adequate data.

IMPACTS ON MAINTENANCE DREDGING

The majority of existing sediment loading settles in the reach between 16th Avenue South and the head of navigation. Construction of the mid-sized channel is not expected to alter the location of this major shoaling area or change the minor shoaling areas. The reach between 16th Avenue South and the head of navigation will be larger and deeper than existing conditions which will cause most of the material to continue to settle in this reach. The location of the major shoaling area may move downstream slightly in the future, however, due to increased flows in the Duwamish River due to operation of the Black River pump station at full capacity.

REFERENCES

Baumgartner, D.J., D.W. Schultz, and J.B. Carkin. 1978. Aquatic Field Investigations, Duwamish Waterway Disposal Site, Puget Sound, Washington; Appendix D: Chemical and Physical Analyses of Water and Sediment in Relation to Disposal of Dredged Material in Elliott Bay. Dredged Material Research Program, U.S. Army Corps of Engineers; Tech. Rep. D-77-24; Vol. 1.

Bernhardt, J. 1980. Duwamish River Intensive Water Quality Survey. Washington St. Department of Ecology. Draft Report; 24 pp.

Brooks, A.S. and N.E. Liptak. 1978. The effect of intermittent chlorination on freshwater phytoplankton. Water Res. 13:49-52.

Brown and Caldwell. 1979. "Combined Sewer Overflow Control Program". Brown and Caldwell Engineers, Seattle, WA. 200+ pp.

Brown and Caldwell, Jones and Jones. 1980. "Preliminary Wastewater Management Plan for the Lake Washington/Green River Basins. Brown and Caldwell Engineers, Seattle, WA; 124 pp.

Dawson, W. A. and L. J. Tilley. 1972. Measurement of salt wedge excursion distance in the Duwamish River Estuary, Seattle, Washington, by means of the dissolved-oxygen gradient. U.S. Geological Survey Water-Supply Paper 1873-D. 27 pp.

Feely, R. A. 1980. "Preliminary Data on the Flocculation of Some Major and Minor Elements in the Duwamish River Estuary". NOAA-PMEL, Seattle, WA. Manuscript: 11 pp.

Fischer, H.B. 1968. Methods for predicting dispersion coefficients in natural streams, with applications to lower reaches of the Green and Duwamish River, Washington. U. S. Geol. Survey Prof. Paper 582-A. 27 pp.

Harper, M. E. 1972. Development and application of a multi-parametric mathematical model of water quality. Ph.D. Thesis. University of Washington, Seattle. 136 pp.

Haushild, W. L. and Stoner, J.D. 1973. Predicted effects of proposed navigation improvements on residence time and dissolved oxygen of the salt wedge in the Duwamish River Estuary, King County, Washington. USGS Open File Report. 13 pp.

Issac, G.W., G.D. Farris, and C.V. Gibbs. 1964. Special Duwamish River studies. Municipality of Metropolitan Seattle, Seattle, WA. 35 pp.

King, D.L. 1970. The role of carbon in eutrophication. Jour. Wat. Poll. Contr. Fed. 42 (12):2035-2051.

Laws, D.A. and T.T. Bannister. 1980. Nutrient-and light-limited growth of Thalassiosira fluviatilis in continuous culture, with implications for phytoplankton growth in the ocean. Limnology and Oceanography; 25 (3):457-473.

National Oceanic and Atmospheric Administration. 1980. Nautical Chart of Seattle Harbor: Elliott Bay and Duwamish Waterway. National Ocean Survey. Chart 14850, 9th Edition.

Odum, H.T. 1956. Primary production in flowing waters. Limnology and Oceanography; 1: 102-117.

Pavlou, S.P. and R.N. Dexter. 1979. Distribution of polychlorinated biphenyls (PCB) in estuarine ecosystems: Testing the concept of equilibrium partitioning in the marine environment. Env. Sci. Tech. 13(1): 65-72.

Prych, E.A., W. L. Haushild, and J.D. Stoner. 1976. Numerical model of the salt-wedge reach of the Duwamish River Estuary, King County, Washington. U.S. Geol. Survey Prof. Paper 990. 34 pp.

Santos, J.F. and J.D. Stoner. 1972. Physical, chemical and biological aspects of the Duwamish River Estuary, King County, Washington, 1963-67. U.S. Geological Survey Water-Supply Paper 1873-C. 74 pp.

Stevens, Thompson and Runyan, Inc. 1972. "Effect of Dredging on Water Quality and Sediment Transport in the Duwamish Estuary". STR, Inc., Seattle, WA. 88 pp.

Stevens, Thompson and Runyan, Inc. September, 1973. "Existing Water Quality and Deficiencies, Duwamish River". RIBCO Task 4360. STR, Inc., Seattle, WA.

Stoner, J.D. 1967. Prediction of salt-water intrusion in the Duwamish River Estuary, King County, Washington. U.S. Geological Survey Prof. Paper 575-D. pp. D253-D255.

Stoner, J.D. 1972. Determination of mass balance and entrainment in the stratified Duwamish River Estuary, King County, Washington. U.S. Geological Survey Water-Supply Paper 1873-F. 17 pp.

Stoner, J.D., W. L. Haushild, and J.B. McConnell. 1975. Numerical model of material transport in salt-wedge estuaries. Part II. Model computation of salinity and salt-wedge dissolved oxygen in the Duwamish River Estuary, King County, Washington. U.S. Geological Survey Professional Paper 917. pp. 13 - 36.

Tilley, L.J. and W.A. Dawson. 1971. Plant nutrients and the estuary mechanism in the Duwamish River Estuary, Seattle, Washington. U.S. Geological Survey Professional Paper 750-C. pp. C185 - C191.

Tilley, L.J. and W. L. Haushild. 1975. Net primary productivity of periphytic algae in the intertidal zone, Duwamish River Estuary, Washington. Jour. Research U.S. Geol. Survey 3(3):253-259.

U.S. Environmental Protection Agency. 1980. Wastewater management plan for the Lake Washington/Green River Basins. Draft EIS. U.S.E.P.A., Seattle, WA. 263 pp.

Washington State Department of Ecology. 1980. Instream resources protection program - Green-Duwamish River Basin, Water Resource Inventory Area (WRIA) 9. Proposed Administrative Rules. Wash. Dept. of Ecology, Olympia. 13 pp.

Welch, E.B. 1969. Factors initiating phytoplankton blooms and resulting effects on dissolved oxygen in Duwamish River estuary, Seattle, Washington. Geological Survey Water-Supply Paper 1873-A. 62 pp.

Welch, E.B., R.M. Emery, R.I. Matsuda, and W.A. Dawson. 1972. The relation of periphytic and planktonic algal growth in an estuary to hydrographic factors. Limnology and Oceanography. 17(s):731-737.

Welch, E.B. and W.T. Trial. 1979. Ammonia toxicity affected by pH and nitrification in the Duwamish River estuary. Report to Brown and Caldwell Engineers, Seattle, Washington.

Winter, D.F., K. Banse, and G.C. Anderson. 1975. The dynamics of phytoplankton blooms in Puget Sound, a fjord in the northwestern United States. Marine Biology, 29:139-176.

Yake, W. 1980. The impact of effluent from the Renton Wastewater Treatment Plant on the dissolved oxygen regime of the Lower Green/Duwamish River. Washington State Department of Ecology, Olympia. 19 pp.

Zison, S.W., W.B. Mills, D. Deimer, and C.W. Chen. 1978. Rates, constants, and kinetics formulations in surface water quality modeling. Tetra Tech Inc., Ca. 317 pp.

APPENDIX: DATA SOURCES

	Data	Year	Source
Physical Characteristics:			
1)	Duwamish Waterway morphometric characteristics	1978	U.S.A.C.E. Condition Surveys
2)	Tidal stage records for Elliot Bay	1966-1976	U.S. Coast and Geodetic Surveys
3)	Air temperature and wind speed at Sea-Tac airport	1967	National Climatic Center, Local Climatological Data
4)	Solar radiation data for km 12.6	1967	Robert Matsuda, Metro, Seattle
5)	Daily discharge records for Green River at Auburn and Tukwila	1961-1979	Water Resources Data for Washington, USGS
6)	Daily discharge records for RTP	1967-1979	Robert Matsuda, Metro, Seattle
7)	Velocity data at selected times and stations in the Duwamish; used in several USGS reports	1966-1967	Edmund Prych, USGS, Tacoma
Water Quality Characteristics:			
1)	Suspended sediment concentrations and loads	1963-1966	Water Resources Data for Washington, USGS
2)	Water quality data for Green River at 212th St. near Kent. Parameters include specific conductance, temp., pH, D.O., bicarbonate, alkalinity, ammonia, kjeldahl nitrogen and total phosphorus	1971, 1974-1976	Water Resources Data for Washington, USGS
3)	Chemical and biological data for the Duwamish, including chlorophyll a and cell count data for km 21, 12.6, 10.4, 7.7, 5.6 and 1.9	1966-1968	Edmund Prych, USGS Tacoma

	Data	Year	Source
4)	Automatic monitor data for Green River-Duwamish (1): Daily summaries of temp., salinity, D.O. and pH for surface monitors at km 21 and 7.7. No pH at bottom monitor at km 7.7 (some data gaps, weekly calibration)	1967-1980	Robert Matsuda, Metro Seattle
5)	Automatic monitor data for Green River-Duwamish (2): Daily summaries of temp., salinity, D.O. and pH for surface monitor at km 1.9. No pH at bottom monitor (some data gaps, weekly calibration).	1967-1976	Robert Matsuda, Metro Seattle
6)	Chemical data for Duwamish at km 21.0, 12.6, 7.7 and 1.9 including Kjeldahl nitrogen, ammonia, nitrate, soluble and total phosphorus. Principally surface samples taken during June-Septemter $TRCl_2$ at station below RTP outfall. SOD measurements during August, 1973.	1966-1980	Robert Matusda, Metro Seattle
7)	RTP effluent characteristics including CBOD, NBOD, ammonia, and $TRCL_2$	1967-1979	Robert Matsuda, Metro Seattle
8)	Chl a, ammonia, nitrate, Total-N, specific conductance, pH and nitrification rates for km 21.0, 20.5, 16.2, 13.2, 10.4, 7.7 and 5.6, Depth profile sampling	1979-1980	Eugene Welch and Wally Trial, Dept. of Civil Eng., UW, Seattle
9)	Data from two Duwamish drift floating surveys above and below RTP. Parameters included temperature, specific conductance, D.O., pH, nitrate, ammonia and $TRCl_2$.	1979	John Bernhardt, DOE, Olympia
10)	Duwamish sediment quality data. Parameters included VSS, COD and Pb.	1969-1980	Ron Thom, USACE Seattle STR (1973)

In addition to the above data sources, compiled data is also available in many of the publications and reports listed in "References".

APPENDIX: CALCULATION OF LOW FLOWS FOR EVALUATION OF PROJECT IMPACTS

1. Renton Treatment Plant
   Existing 1980 dry weather flow = 56 cfs
   Future 2000 dry weather flow = 112 cfs
   Metro estimates elimination of discharge to Green River by 1987; flow = 82 cfs

2. Existing Green River at Tukwila flows
   (Refer to Table A-1)
   Minimum daily flow since 1961 = 198 cfs
   Average minimum daily flow = 267 cfs
   Minimum monthly flow in summer = 274 cfs (Sept. 1967)

3. Department of Ecology Minimum Flows - Auburn gage
   For gage near Auburn from July 15 through October 15 of normal water years, minimum flows = 300 cfs

4. Contribution of local drainage runoff between Auburn and Tukwila gages = 44 cfs average (Table A-2)

5.a. Existing minimum flows prior to DOE minimum flows:

Use minimum monthly flow at Renton =	274 cfs
plus existing Renton TP flows =	56 cfs
	330 cfs

b. Existing condition, using DOE minimum flows:

DOE minimum flow near Augurn =	300 cfs
plus local tributary drainage =	44 cfs
plus Renton TP flow =	56 cfs
	400 cfs

c. Future condition - 1987

Green River at Tukwila =	344 cfs
plus Renton TP flow =	82 cfs
	426 cfs

d. Ultimate flow condition - no Renton TP discharge

Green River at Tukwila =	344 cfs

e. Summary: range in flows = 330 - 426 cfs
   = 9.3 - 12.1 m^3/sec.

TABLE A-1

Selected Flow Characteristics of Green River at Tukwila
(all flows are in cfs)

Water Year	Ave. Annual Discharge	Daily Minimum Q	Daily Minimum Date	Average Monthly Flows June	July	Aug.	Sept.
1961	1,761	198	9-14-61	1,112	444	235	242
1962	-	-	-	924	548	363	354
1963	1,314	269	9-30-63	686	587	373	306
1964	1,765	251	10- 2-63	2,933	1,129	569	730
1965	1,682	260	9-30-65	687	360	316	276
1966	1,058	264	10- 1-65	827	743	320	279
1967	1,512	257	9-28-67	955	371	331	274
1968	1,500	273	8-12-68	1,222	440	448	936
1969	1,607	231	9-15-69	1,439	711	341	365
1970	1,255	261	8-31-70	860	387	289	357
1971	1,750	287	10- 4-70	1,478	1,023	416	422
1972	2,312	330	10-13-71	1,734	1,207	502	670
1973	1,107	250	9-12-73	468	511	312	341
1974	2,082	300	9-26-74	3,050	1,056	561	417
1975	1,628	250	10-20-74	1,524	781	500	499
1976	2,028	305	10- 1-75	1,005	622	495	573
1977	813	250	8-17-77	860	331	299	583
1978	1,517	321	8-10-78	752	467	353	661
1979	1,193	250	9-19-79	454	334	294	294
Averages	1,371	267		1,214	634	385	452

TABLE A-2

Estimate of Local Inflows to Green River

Between Auburn Gage and Tukwila Gage

Water Year	Month	Monthly Discharge (cfs) Auburn	Tukwila	Increase
1963	September	276	306	30
1964	September	696	730	34
1965	September	240	276	36
1966	September	263	279	16
1967	September	237	274	37
1968	July	370	440	70
1969	August	262	341	79
1970	August	247	289	42
1971	September	376	422	46
1972	August	409	502	93
1973	August	257	312	55
1974	September	386	417	31
1975	September	469	499	30
1976	August	462	495	33
1977	August	257	299	42
1978	August	306	353	47
1979	August	261	294	33
			Average =	44 cfs

DATE
ILMED
-8